Final

Baseline Ecological Risk Assessment Problem Formulation and Study Design for the Baby Bains Gap Road Ranges

Fort McClellan
Calhoun County, Alabama

Prepared For:

U.S. Army Corps of Engineers
Mobile District
109 St. Joseph Street
Mobile, Alabama 36602

Prepared By:

Shaw Environmental, Inc.
312 Directors Drive
Knoxville, Tennessee 37923

Task Order CK11
Contract No. DACA21-96-D-0018
Shaw Project No. 800486

August 2006

Table of Contents

Page

Table of Contents *(Continued)*

Page

Table of Contents (Continued)

Page

List of Tables

List of Figures

1.0 Introduction

A screening-level ecological risk assessment (SLERA) was conducted for the Baby Bains Gap Road (BBGR) Ranges and presented in the report entitled *Remedial Investigation Report, Baby Bains Gap Road Ranges* (Shaw Environmental, Inc. [Shaw], 2004a). The results of the SLERA indicated that several constituents in environmental media at the BBGR Ranges have the potential to pose adverse ecological risks. Therefore, a Baseline Ecological Risk Assessment (BERA) will be completed for the BBGR Ranges in order to reduce the level of uncertainty inherent in the SLERA process and to better define the potential for ecological risks. Per the U.S. Environmental Protection Agency's (EPA) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997), the first step in the BERA process is the "Problem Formulation." The Problem Formulation also constitutes "Step 3" of the EPA's eight-step process (EPA, 1997).

The Problem Formulation for the BBGR Ranges uses the results of the SLERA and on-site reconnaissance to identify the specific ecological values to be protected at the BBGR Ranges, which are then used to establish assessment endpoints. The questions and issues that need to be addressed in the BERA are also defined in this Problem Formulation.

The Problem Formation phase of the BERA addresses and expounds upon a number of issues described in the SLERA, including:

- Refinement of the constituents of potential ecological concern (COPEC) identified in the SLERA
- Description of the ecotoxicity of the COPECs
- Description of the fate and transport of the COPECs
- Description of the ecosystems potentially at risk
- Development and refinement of the site conceptual model
- Refinement of the complete exposure pathways
- Identification of the assessment endpoints.

Also included in this report is the BERA Study Design. The Study Design for the BBGR Ranges utilizes the information gathered and presented in the Problem Formulation to establish measurement endpoints and to design studies that are appropriate to test the hypotheses concerning the assessment endpoints. Data quality objectives as well as statistical approaches are also presented in the BERA Study Design.

2.0 Identification of Constituents of Potential Ecological Concern

The SLERA for the BBGR Ranges initially identified a number of COPECs in soil as well as in sediments in South Branch of Cane Creek and Ingram Creek tributaries. COPECs were initially identified by calculating screening-level hazard quotients, which were developed via a three-step process as follows:

- Comparison of maximum detected constituent concentrations to ecological screening values (ESV)
- Identification of essential macro-nutrients
- Comparison to naturally occurring background concentrations.

Constituents that were detected in environmental media at the BBGR Ranges were evaluated against the ESVs by calculating a screening-level hazard quotient (HQ_{screen}) for each constituent in each environmental medium. An HQ_{screen} was calculated by dividing the maximum detected constituent concentration in each environmental medium by its corresponding ESV as follows:

$$HQ_{screen} = \frac{MDCC}{ESV}$$

where:

HQ_{screen}	=	screening-level hazard quotient
$MDCC$	=	maximum detected constituent concentration
ESV	=	ecological screening value.

A calculated HQ_{screen} value of one or less indicated that the maximum detected constituent concentration (MDCC) was equal to or less than the chemical's conservative ESV, and was interpreted in the SLERA as a constituent that does not pose a potential for adverse ecological risk. Conversely, an HQ_{screen} value greater than one indicated that the MDCC was greater than the ESV and that the chemical might pose adverse ecological hazards to one or more receptors.

In order to better understand the potential risks posed by chemical constituents at the BBGR Ranges, a mean hazard quotient was also calculated in the SLERA by comparing the arithmetic mean constituent concentration in each environmental medium to the corresponding ESV. The calculated screening-level hazard quotients for surface soil, surface water, sediment, and groundwater at the BBGR Ranges are presented in Tables 2-1 through 2-11.

EPA recognizes several constituents in abiotic media that are necessary to maintain normal function in many organisms. These essential macronutrients are iron, magnesium, calcium, potassium, and sodium (EPA, 1989). Most organisms have mechanisms designed to regulate nutrient fluxes within their systems; therefore, these nutrients are generally only toxic at very high concentrations. Although iron is an essential nutrient and is regulated within many organisms, it may become increasingly bioavailable at lower soil pH values, thus increasing its potential to elicit adverse affects. Therefore, iron was not evaluated as an essential nutrient in the SLERA. Essential macronutrients were only considered COPECs if they were present in site samples at concentrations ten times the naturally occurring background concentration.

The comparison of detected constituent concentrations with naturally occurring constituent concentrations was conducted via a three-tier process outlined in a technical memorandum dated March 14, 2005 (Shaw, 2005). The first tier of the background comparison process was a comparison of the maximum detected constituent concentration to the background threshold value (BTV). A study of the natural geochemical composition associated with FTMC (SAIC, 1998) determined the mean concentrations of 24 metals in surface soil, surface water, sediment, and groundwater samples collected from presumably un-impacted areas. Per agreement with EPA Region 4, the BTV for each metal was calculated as two times the mean background concentration for that metal. The BTV for each metal was used to represent the upper boundary of the range of natural background concentrations expected at FTMC, and was used as the basis for evaluating metal concentrations measured in site samples. Site sample metal concentrations less than or equal to the corresponding BTV represent the natural geochemical composition of media at FTMC, and not contamination associated with site activity. Site sample metal concentrations greater than the corresponding BTV require further background assessment.

If maximum constituent concentrations were greater than the BTV, then the second tier of the background comparison was employed. Tier two of the background comparison consists of statistical comparisons of the site data to background data using the Slippage Test and the Wilcoxon Rank Sum (WRS) Test. If the site data failed either the Slippage Test or the WRS Test, then the site data were subjected to a geochemical evaluation (Tier 3) to determine whether concentrations of inorganic compounds are naturally occurring or are elevated due to contamination.

Thus, the first step in determining screening-level hazard quotients was a comparison of maximum detected constituent concentrations to appropriate ESVs. Constituents with HQ_{screen} values less than one were considered to pose insignificant ecological risk and were eliminated from further consideration. Constituents with HQ_{screen} values greater than one were eliminated from further consideration if they were macro-nutrients and their detected concentrations were less than 10-times naturally occurring levels. Those constituents that had HQ_{screen} values greater one and were not considered macro-nutrients were then compared to background using the three-tier background screening process. If constituent concentrations were determined to be less than their naturally occurring background concentrations, then a risk management decision could result in eliminating these constituents from further assessment.

Thus, constituents were initially identified as COPECs within the SLERA if all of the following conditions were met:

- The maximum detected constituent concentration exceeded the ESV
- The maximum detected constituent concentration was 10-times the BTV if constituent is a macronutrient
- The maximum detected constituent concentration exceeded the BTV for inorganics.

If a constituent did not meet all of these conditions, then it was not considered a COPEC at the BBGR Ranges and was not considered for further assessment. Again, identification of a constituent as a COPEC in the SLERA simply indicated that further assessment of that particular constituent was deemed appropriate but did not imply that a particular constituent posed a definite risk to ecological receptors.

In order to focus the BERA efforts on the constituents that are the most prevalent at the BBGR Ranges and have the greatest potential to pose ecological risk, additional lines of evidence were assessed in the SLERA to refine the initial list of COPECs. These additional lines of evidence were scrutinized to aid in the decision process of whether or not to include a constituent as a COPEC in the BERA for the BBGR Ranges. These additional lines of evidence are discussed in the following sections.

2.1 COPECs in Surface Soil

Antimony, copper, and lead were frequently detected in surface soil at virtually all of the BBGR Ranges at concentrations that exceeded their respective ESVs and naturally occurring levels. The highest concentrations of these three constituents were found in locations that would be anticipated based on the site usage as firing ranges (i.e., soil berms that act as impact areas). Thus, it could be concluded that these constituents are site-related and could be considered COPECs in surface soil at virtually all of the BBGR Ranges.

2.1.1 Range 18

Antimony, copper, and lead were frequently detected at elevated concentrations in surface soil at Range 18. The highest concentrations of copper and lead were found in soil samples collected from the hillside in the southern portion of the range, which acted as an impact zone for Range 18.

Antimony. All of the detected concentrations of antimony (7 out of 16 samples) in surface soil at Range 18 exceeded the ESV and BTV. Additionally, all of the detected antimony concentrations in soil were greater than the background range for antimony. Because antimony is a known component of small-arms ammunition and was routinely detected at elevated concentrations, it was identified as a COPEC in surface soil at Range 18.

Arsenic. Arsenic was detected in three out of 16 surface soil samples at concentrations that exceeded the BTV. All of the soil samples from Range 18 exhibited arsenic concentrations that were within the range of background. Arsenic passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected arsenic in surface soil from Range 18 was determined to be naturally occurring through geochemical evaluation except for one sample. The calculated HQ_{screen} value was 2.4, indicating the maximum detected concentration of arsenic in surface soil only slightly exceeded the ESV. Based on the relative infrequency of elevated arsenic detections, the low magnitude of the HQ_{screen} value, and the magnitude of the detected concentrations relative to background, arsenic was not considered a COPEC in surface soil at Range 18.

Beryllium. Beryllium was detected in five out of 16 samples at concentrations that exceeded the ESV. Six samples also exceeded the range of background. Geochemical analysis indicated that three samples had anomalously high concentrations of beryllium, which may indicate a component of contamination. All of the elevated concentrations of beryllium were from samples located in the hillside in the southern portion of Range 18 that served as the impact area for this range. For these reasons, beryllium was identified as a COPEC in surface soil at Range 18.

Copper. Copper was detected in nine out of 16 samples at concentrations that exceeded the ESV. Eleven samples exhibited copper concentrations that were greater than the background range. The highest concentrations of copper in surface soil were detected in samples collected from the hillside impact area in the southern portion of the range. For these reasons, copper was identified as a COPEC in surface soil at Range 18.

Lead. Lead was detected in 33 out of 55 surface soil samples at concentrations that exceeded the ESV. Twenty-nine samples exhibited lead concentrations that were greater than the background range. The highest detected concentrations of lead were found in samples from the hillside impact area located in the southern portion of Range 18. Lead is a known component of small-arms ammunition and the historical use of the site as a firing range indicates that the lead in soil is related to Army activities. For these reasons, lead was identified as a COPEC in surface soil at Range 18.

Nickel. Nickel was detected in two out of 16 surface soil samples at concentrations that exceeded the ESV. There was no discernable pattern to the elevated nickel concentrations in surface soil. Based on geochemical evaluation, only one surface soil sample at Range 18 exhibited an anomalously high concentration of nickel. The calculated HQ_{screen} value was 2.2, indicating the maximum detected concentration of nickel only slightly exceeded the ESV. Based on the infrequency of detection and the low magnitude of the HQ_{screen} value, nickel was not considered a COPEC in surface soil at Range 18.

Zinc. Zinc was detected in eight out of 16 surface soil samples at concentrations that exceeded the ESV. Three samples exhibited zinc concentrations that were greater than the background range. Geochemical analysis indicated that two surface soil samples had anomalously high concentrations of zinc, which may indicate a component of contamination. The highest detected concentrations of zinc were found in samples from the hillside impact area located in the southern portion of Range 18. Zinc is a known component of ammunition and the historical use of the site as a firing range indicates that the zinc in soil is related to Army activities. For these reasons, zinc was identified as a COPEC in surface soil at Range 18.

Organic Compounds. The herbicide MCPP was detected in two out of 11 surface soil samples at concentrations that exceeded the ESV. Studies of the fate and transport of MCPP have shown that the duration of MCPP's residual activity in soil is about two months. Because this range has been inactive for a number of years, it is expected that the MCPP detected in soil no longer exhibits any residual activity. Additionally, MCPP has been shown to be practically nontoxic to birds (LC_{50} > 5,620 ppm for mallards and LC_{50} > 5,000 ppm for bobwhite quail) and nontoxic to fish (LC_{50} =124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (Extension Toxicology Network [EXTOXNET], 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, the detected concentrations of MCPP present a low potential for toxicity, MCPP does not have the propensity to bioaccumulate, and it has aged sufficiently to render it inactive, MCPP was not considered a COPEC in surface soil at Range 18.

The pesticides 4,4'-DDT, beta-BHC, and endrin ketone were all detected in one sample at concentrations that exceeded the ESV. The calculated HQ_{screen} values for these pesticides were 1.8, 1.5, and 1.9, respectively, indicating the maximum detected concentrations of these pesticides only slightly exceed their ESVs. Based on their infrequency of detection and low HQ_{screen} values, these pesticides were not considered COPECs in surface soil at Range 18.

Several semivolatile organic compounds (SVOC) were detected in surface soil at concentrations that exceeded their respective ESVs, namely 2,4-dinitrotoluene, anthracene, benzo(a)pyrene, fluoranthene, and pyrene. 2,4-Dinitrotoluene was detected in one out of 18 samples, anthracene was detected in one out of 11 samples, benzo(a)pyrene was detected in one out of 11 samples, fluoranthene was detected in two out of 11 samples, and pyrene was detected in two out of 11 samples at elevated concentrations. The calculated HQ_{screen} values for these SVOCs were 1.2, 2.1, 4.4, 4.3, and 4.0, respectively. These SVOCs were detected in the same two samples, neither of which is associated with areas that would be indicative of munitions-related activities. The polycyclic aromatic hydrocarbon (PAH) background screening values for soil adjacent to asphalt (IT Corporation [IT], 2000) are 0.935 mg/kg for anthracene, 1.42 mg/kg for benzo(a)pyrene, 2.031 mg/kg for fluoranthene, and 1.626 mg/kg for pyrene. All of the detected concentrations for these four PAHs were less than these screening values. Based on the infrequency of detection, the low magnitude of the HQ_{screen} values, and the detected concentrations relative to the PAH screening values, none of these SVOCs were considered COPECs in soil at Range 18.

2.1.2 Range 20

Copper and lead were frequently detected at elevated concentrations in surface soil at Range 20. The highest concentrations of copper and lead were found in soil samples collected from the hillside opposite the firing line which acted as the impact zone for Range 20.

Antimony. Antimony was detected in two out of four surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 2.9. Although the HQ_{screen} value did not indicate a severe exceedance of the ESV, antimony is a known component of ammunition and the highest concentrations of antimony were found in the impact area for this range. Because antimony is a known component of ammunition and the highest concentrations were detected in areas that would suggest it was the result of Army activities, antimony was considered a COPEC in surface soil at Range 20.

Copper. Copper was detected in two out of four surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 6.2. The highest concentrations of copper were detected in samples from the hillside impact area for this range. Based on these reasons and because it is a known component of ammunition, copper was identified as a COPEC in surface soil at Range 20.

Lead. Lead was detected in 14 out of 15 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited lead concentrations that were greater than the background range. The highest concentrations of lead were detected in samples from the hillside impact area for this range. Based on these reasons and the fact that lead is a known component of ammunition, lead was identified as a COPEC in surface soil at Range 20.

2.1.3 Range 23

Copper and lead were detected frequently in surface soil samples from Range 23 at elevated concentrations. The highest concentrations of these constituents were found in the target mounds located at various distances downrange of the firing lines.

Antimony. Antimony was detected in three out of 22 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 2.4, indicating the maximum detected concentration of antimony only slightly exceeded the ESV. However, because it is a known component of ammunition, antimony was considered a COPEC in surface soil at Range 23.

Copper. Copper was detected in 11 out of 22 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited copper concentrations that were greater than the range of background for copper. The highest concentrations of copper in surface soil were detected in the vicinity of the target mounds along each firing line. Based on its frequency and pattern of detection and because it is a known component of ammunition, copper was identified as a COPEC in surface soil at Range 23.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and BTV. The highest concentrations of lead were also detected in the vicinity of the target mounds along each of the firing lines. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and because it is a known component of ammunition, lead was considered a COPEC in surface soil at Range 23.

Organic Compounds. Aroclor 1260 was detected in one out of 13 surface soil samples. The calculated HQ_{screen} value was 1.1, indicating the maximum detected concentration of Aroclor-

1260 only slightly exceeded the ESV. Based on its infrequency of detection and low HQ_{screen} value, Aroclor-1260 was not considered a COPEC in surface soil at Range 23. Several pesticides exceeded their respective ESVs in one or more surface soil samples, namely 4,4'-DDE, 4,4'-DDT, aldrin, beta-BHC, dieldrin, endrin, gamma-BHC, and methoxychlor. 4,4'-DDE and 4,4'-DDT were detected in one out of 13 samples at elevated concentrations, with HQ_{screen} values of 1.3 and 1.2, respectively. Aldrin was detected in two out of 13 samples at elevated concentrations, with an HQ_{screen} value of 5.2. Beta-BHC and dieldrin were detected in three out of 13 samples at elevated concentrations, with HQ_{screen} values of 6.6 and 7.8, respectively. Endrin was detected in one out of 13 samples at an elevated concentration, with an HQ_{screen} value of 3.6. Gamma-BHC was detected in two out of 13 samples at elevated concentrations, with an HQ_{screen} value of 420. Methoxychlor was detected in one out of 13 samples at an elevated concentration, with an HQ_{screen} value of 1.1. There was no apparent pattern to the elevated concentrations of pesticides. Based on the relative infrequency of detection, the lack of an apparent pattern to the detections, and the overall relatively low HQ_{screen} values, none of the detected pesticides were included as COPECs in surface soil at Range 23.

The herbicide MCPP was detected in three out of 13 surface soil samples at elevated concentrations. Studies of the fate and transport of MCPP have shown that the duration of MCPP's residual activity in soil is about two months. Because this range has not been active for a number of years, it is expected that the MCPP detected in soil no longer exhibits any residual activity. Additionally, MCPP has been shown to be practically nontoxic to birds (LC_{50} > 5,620 parts per million [ppm] for mallards and LC_{50} >5,000 ppm for bobwhite quail) and nontoxic to fish (LC_{50} =124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (EXTOXNET, 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, the detected concentrations of MCPP present a low potential for toxicity, MCPP does not have the propensity to bioaccumulate, and it has aged sufficiently to render it inactive, MCPP was not considered a COPEC in surface soil at Range 23.

Trichloroethene, 2,4-DNT, and 2,6-DNT were detected in one of 13 samples at elevated concentrations. The calculated HQ_{screen} values for these constituents were 1.8, 1.5, and 2.6, respectively. Based on their infrequency of detection and low HQ_{screen} values, these constituents were not considered COPECs in soil at Range 23.

2.1.4 Range 25

Antimony, copper, and lead were detected frequently at elevated concentrations in surface soil samples from Range 25. The highest concentrations of these constituents were found in the hillside impact zone for this range.

Antimony. Antimony was detected in nine out of 62 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 5.9. There was no apparent pattern to the elevated antimony concentrations. However, because it is a known component of ammunition, antimony was considered a COPEC in surface soil at Range 25.

Copper. Copper was detected in 13 out of 62 surface soil samples at concentrations that exceeded the ESV. Twenty samples exhibited copper concentrations that were greater than the range of background copper. The highest concentrations of copper in surface soil were detected in the hillside that acted as the impact zone for this range. Based on the frequency of detection, the pattern of detection, and because it is a known component of ammunition, copper was identified as a COPEC in surface soil at Range 25.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. The highest concentrations of lead were also detected in the hillside impact zone for this range. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and because it is a known component of ammunition, lead was considered a COPEC in surface soil at Range 25.

Zinc. Zinc was detected in 19 out of 62 surface soil samples from Range 25 at elevated concentrations. Geochemical evaluation indicated one surface soil sample exhibited an elevated concentration of zinc compared to naturally occurring levels. All other zinc concentrations were determined to be naturally occurring. The calculated HQ_{screen} value was 4.4. Based on the infrequency of detection above naturally occurring concentrations, zinc was not considered a COPEC in surface soil at Range 25.

Organic Compounds. 4,4'-DDT was detected two out of 12 samples at elevated concentrations relative to the ESV. The calculated HQ_{screen} value was 2.3. Based on the relative infrequency of detection at elevated concentrations and the low magnitude of the HQ_{screen} value, 4,4'-DDT was not considered a COPEC in soil at Range 25.

2.1.5 Range 25-East

No constituents were detected in surface soil samples from Range 25-East at elevated concentrations; therefore, no COPECs have been identified in surface soil at this range.

2.1.6 Range 26

Antimony, copper, and lead were detected frequently at elevated concentrations in surface soil samples from Range 26. The highest concentrations of copper and lead were found in the hillside impact zone for this range.

Antimony. Antimony was detected in four out of 17 surface soil samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value was 4.4. There was no apparent pattern to the elevated antimony detections. However, because it is a known component of ammunition, antimony was included as a COPEC in surface soil at Range 26.

Copper. Copper was detected in 4 out of 19 surface soil samples at concentrations that exceeded the ESV. Nine samples exhibited copper concentrations that were greater than the range of background for copper. The highest concentrations of copper in surface soil were detected in the hillside impact zone for this range. Based on its frequency and pattern of detection and because it is a known component of ammunition, copper was identified as a COPEC in surface soil at Range 26.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. The highest concentrations of lead were also detected in the hillside impact zone for this range. Based on the frequency of detection at elevated concentrations, the pattern of elevated concentrations consistent with the firing line configuration, and because it is a known component of ammunition, lead was considered a COPEC in surface soil at Range 26.

Organic Compounds. Several semi-volatile organic compounds (SVOC) were detected at elevated concentrations in surface soil at Range 26. These SVOCs include: anthracene, benzo(a)pyrene, fluoranthene, pentachlorophenol, phenanthrene, and pyrene. Anthracene was detected in one sample out of 11 at an elevated concentration. The calculated HQ_{screen} value for anthracene was 5.2. Benzo(a)pyrene was detected in two out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for benzo(a)pyrene was 25. Fluoranthene was detected in four out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for fluoranthene was 58. Pentachlorophenol was detected in one out of 11 samples at an elevated concentration. The calculated HQ_{screen} value for pentachlorophenol was 40.5. Phenanthrene was detected in one out of 11 samples at an elevated concentration. The calculated HQ_{screen} value for phenanthrene was 2.4. Pyrene was detected in three out of 11 samples at elevated concentrations. The calculated HQ_{screen} value for phenanthrene was 50. There was no apparent pattern to the elevated concentrations of SVOCs. Due to the relative infrequency of detection and lack of discernable pattern relative to Army activities, SVOCs were not considered COPECs in surface soil at Range 26.

The pesticides beta-BHC, dieldrin, endosulfan sulfate, and MCPP were each detected in 1 of 11 surface soil samples at elevated concentrations. The calculated HQ_{screen} values for these compounds were 5.3, 1.02, 1.2, and 13. There was no discernable pattern to the elevated concentrations of these pesticides. Due to the relative infrequency of detection, the low magnitude of the HQ_{screen} values, and the lack of discernable pattern relative to Army activities, these pesticides were not considered COPECs in surface soil at Range 26

2.1.7 Ranges South of Range 25

Antimony, copper, and lead were detected frequently at elevated concentrations in surface soil samples from the Ranges South of Range 25.

Antimony. Antimony was detected at elevated concentrations in nine out of 17 surface soil samples. The calculated HQ_{screen} value was 2.1. Because it is a known component of ammunition and was detected frequently at elevated concentrations, antimony was identified as a COPEC in surface soil at the Ranges South of Range 25.

Copper. Copper was detected in 9 out of 16 surface soil samples at concentrations that exceeded the ESV. Twelve samples exhibited copper concentrations that were greater than the range of background copper. Based on the frequency of detection at elevated concentrations and because it is a known component of ammunition, copper was identified as a COPEC in surface soil at the Ranges South of Range 25.

Lead. Lead was detected frequently at elevated concentrations relative to the ESV and naturally occurring levels. Based on the frequency of detection at elevated concentrations and because it is a known component of ammunition, lead was considered a COPEC in surface soil at the Ranges South of Range 25.

2.2 COPECs in Surface Water

No constituents were detected at elevated concentrations in surface water samples (which were analyzed for total recoverable concentrations) from either South Branch of Cane Creek or

Ingram Creek tributaries. Therefore, no COPECs were identified in surface water at the BBGR Ranges.

2.3 COPECs in Sediment

No constituents were detected at elevated concentrations in sediment samples from the South Branch of Cane Creek tributaries; therefore, no COPECs were identified in South Branch of Cane Creek tributary sediments. Copper and lead were detected in one sample from the Ingram Creek tributaries at anomalously high concentrations relative to naturally occurring levels. These results indicate that these samples may contain a component of contamination. Copper and lead are also known components of ammunition. For these reasons, copper and lead were identified as COPECs in sediment within the Ingram Creek tributaries.

The pesticides 4,4'-DDT, MCPP, alpha-BHC, gamma-chlordane, and heptachlor were detected in sediment samples from Ingram Creek tributaries. 4,4'-DDT was detected in one out of 9 sediment samples. The calculated HQ_{screen} value was 9.7. MCPP was detected in two out of 9 sediment samples from Ingram Creek tributaries. An HQ_{screen} value could not be calculated because MCPP does not have an ESV. Alpha-BHC was detected in one out of 9 sediment samples at an elevated concentration. The calculated HQ_{screen} value for alpha-BHC was 1.4. Gamma-chlordane was detected in two out of 9 sediment samples at elevated concentrations. The calculated HQ_{screen} value for gamma-chlordane was 52.4. Heptachlor was detected in two out of 9 sediment samples at elevated concentrations. The calculated HQ_{screen} value for heptachlor was 2.0.

Studies of the fate and transport of MCPP have shown that the duration of its residual activity in soil or sediment is about two months. Because this range has not been active for a number of years, it is expected that the MCPP detected in sediment no longer exhibits any residual activity. Additionally, MCPP has been shown to be practically nontoxic to birds (LC_{50} > 5,620 ppm for mallards and LC_{50} >5,000 ppm for bobwhite quail) and nontoxic to fish (LC_{50} = 124 ppm for rainbow trout and LC_{50} > 100 ppm for bluegill sunfish) and to bees (EXTOXNET, 2003). MCPP has also been shown to have a low potential to bioaccumulate in fish (EXTOXNET, 2003). Because MCPP was infrequently detected, the detected concentrations of MCPP present a low potential for toxicity, MCPP does not have the propensity to bioaccumulate, and it has aged sufficiently to render it inactive, MCPP was not considered a COPEC in sediment in the Ingram Creek tributaries.

The other pesticides were not considered COPECs in sediment due to their relative infrequency of detection, limited areal extent, and/or low calculated HQ_{screen} values.

Acetone was detected in one out of 9 sediment samples at a slightly elevated concentration. The calculated HQ_{screen} value for acetone in sediment was 1.2. Based on its infrequency of detection at elevated concentrations and the low HQ_{screen} value, acetone was not identified as a COPEC in sediment in the Ingram Creek tributaries.

2.4 COPECs in Groundwater

4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, gamma-BHC, and heptachlor, were each detected in one out of 19 groundwater samples at elevated concentrations. Alpha-chlordane was detected in two out of 19 groundwater samples at elevated concentrations. The 4,4'-DDD, 4,4'-DDT, endosulfan I, endrin, and heptachlor detections were all from the same monitoring well located in the center of Range 25. Gamma-BHC was detected in one monitoring well at Range 18 and alpha-chlordane was detected in one well each at Ranges 23 and 25. The HQ_{screen} values for 4,4'-DDD, 4,4'-DDT, alpha-chlordane, endosulfan I, endrin, gamma-BHC, heptachlor in groundwater were calculated to be 13.9, 77.0, 8.8, 1.4, 23.5, 1.6, and 31.6, respectively.

It is important to note that none of the pesticides detected in groundwater were detected in surface water samples from the BBGR Ranges. Ecological receptors do not have a direct exposure pathway to groundwater. Ecological receptors can only be exposed to constituents in groundwater if groundwater is expressed at the ground surface as seeps or is discharged to lakes or streams via springs. Exposure of ecological receptors to groundwater could then occur via surface water pathways. Contaminants that may have entered groundwater in the past are likely to have been mostly, if not entirely, transported to surface water bodies by now, and if ongoing groundwater contamination of surface water bodies were a concern, surface water samples would indicate the presence of groundwater contaminants. Because the pesticides in groundwater were detected infrequently and were not detected in surface water samples, it is expected that potential ecological exposure to these compounds is insignificant. Therefore, these compounds were not considered COPECs in groundwater at the BBGR Ranges.

2.5 Summary of COPECs

In order to focus this Problem Formulation on the constituents that are most prevalent at the BBGR Ranges and have the greatest potential to pose adverse ecological effects to local ecological communities and populations, the initial list of COPECs was scrutinized using additional lines of evidence. These additional lines of evidence included frequency of detection, magnitude of the HQ_{screen} value, comparison to alternative screening values, association with Army activities, bioaccumulation, and toxicity potential. Based on these additional lines of evidence, the COPECs that have been identified at the BBGR Ranges are summarized below and presented in Table 2-12:

- **Surface Soil:** antimony, beryllium, copper, lead, and zinc
- **Surface Water:** none
- **Sediment:** copper, lead, and gamma-chlordane.

3.0 Ecotoxicity

The ecotoxicological properties of the COPECs identified at the BBGR Ranges dictate which receptors have the greatest potential ecological risk and the pathways by which those receptors have the greatest potential for exposure. Factors such as the propensity to bioaccumulate or biomagnify, as well as their acute and/or chronic toxicity to immature or adult receptors are important factors in the consideration of a constituent's ecotoxicity and also in the development of assessment and measurement endpoints. Current ecological risk assessment methodologies generally address chronic exposures and effects since they generally provide for more ecological protection than methods for assessing acute exposures and effects. Some ecological risk assessment test methodologies (i.e. acute surface water toxicity tests) directly assess acute exposures, and the results are extrapolated to assess chronic exposures. However, a thorough ecological risk assessment addresses both acute and chronic toxicity.

In order for a constituent to exhibit toxicity or to bioaccumulate, it first must be bioavailable. In general, there are three microbial processes affecting the bioavailability of metals (Connell and Miller, 1984). The first is biodegradation of organic matter into lower molecular weight compounds, which are more capable of complexing metal ions than higher molecular weight organic molecules. The second is alterations to physico-chemical properties of metals by microbial metabolic activities (i.e., oxidation-reduction potential and pH conditions). Finally, the process of bacterial methylation, specifically of lead, may greatly enhance the bioavailability of certain inorganic compounds.

The actual uptake of bioavailable metals by terrestrial and aquatic organisms is through three main routes: 1) uptake across respiratory surfaces (lungs or gills), 2) adsorption from soil, sediment or water onto body surfaces, and 3) ingestion of food, water or incidental particles. Given the state of science relative to bio-uptake dynamics, the ingestion route is the most quantifiable uptake route at this time. Metal uptake from dietary sources, in comparison to direct adsorption, is also considered the primary uptake route in small terrestrial and aquatic receptors.

Although ecological receptors can readily absorb metals from food/water ingestion, their ability to regulate elevated concentrations of metals dictates their tolerance and is a critical factor in survival. Once the upper limit, or threshold, of metal sequestration and excretion is reached, sub-lethal effects such as inhibited reproduction and growth potentials may be exhibited, followed by lethality. Temporary metal storage is generally by binding to proteins, such as metallotheioneins, polysaccharides, and amino acids (Connell and Miller, 1984). Storage within liver and kidney tissues as well as bone, feathers, and fur also provide a useful means for sequestering metals such as lead.

Considerable inter- and intra-species differences exist in bioaccumulation potential of individual metals. In addition, according to Phillips (1980), different chemical forms of any one metal may be absorbed and excreted at widely differing rates. Many studies support the premise that inorganic metals do not have a high propensity to biomagnify up through food chains.

The following sections highlight key toxicological properties of the COPECs that have been identified at the BBGR Ranges (antimony, beryllium, copper, lead, gamma-chlordane, and zinc).

3.1 Antimony

Antimony binds to soil and particulates (especially those containing iron, manganese, or aluminum) and is oxidized by bacteria in soil. Exposure routes for mammals include ingestion and inhalation. Antimony does not tend to biomagnify in terrestrial food chains (Ainsworth, 1988), and is not significantly metabolized and excreted in the urine and feces. Antimony at elevated levels has the potential to cause reproductive, pulmonary, and hepatic effects in mammals (EPA, 1999a).

Plants. Antimony is considered a non-essential element and is easily taken up by plants if available in the soil in soluble forms (Kabata-Pendias and Pendias, 1992). A screening level of 5.0 milligrams per kilogram (mg/kg) has been proposed by Kabata-Pendias and Pendias (1992) based on a report of unspecified phytotoxic responses by plants grown in soil amended with antimony.

Terrestrial Invertebrates. EPA (2005a) has developed an ecological soil screening level (eco-SSL) for antimony of 78 mg/kg. The eco-SSL for antimony is the geometric mean of three EC_{20} values reported in the literature. Kuperman et al. (2002) reported an EC_{20} value using enchytraeids (*Enchytraeus crypticus*) of 194 mg/kg; Phillips et al. (2002) reported an EC_{20} value using springtails (*Folsomia candida*) of 81 mg/kg; and Simini et al. (2002) reported an EC_{20} value using earthworms (*Eisenia fetida*) of 30 mg/kg.

Mammals. Female mice exposed to 5.0 milligrams per liter (mg/L) antimony (as antimony potassium tartrate) in their drinking water showed a reduction in their lifespan. This dose was equivalent to a lowest-observed-adverse-effects-level (LOAEL) of 1.25 mg/kg/ per day [mg/kg/day]), which can be converted to a no-observed-adverse-effects-level (NOAEL) of 0.125 mg/kg/day (Integrated Risk Information Service, 2001).

Laboratory data on antimony toxicity (as antimony potassium tartrate) in laboratory mice through drinking water ingestion were used to estimate a chronic NOAEL value of 0.125 mg/kg/day (Schroeder et al., 1968). Lifespan and longevity were the endpoints tested.

EPA (2005a) has derived an Eco-SSL for mammalian wildlife species of 0.27 mg/kg antimony in soil. This mammalian Eco-SSL is the lowest calculated value based on reproduction, growth, and survival of ground insectivores (shrew).

Birds. No information was found regarding the potential toxicity of antimony to birds.

Aquatic Life. The available data for antimony indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 9.0 and 1.6 mg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae can occur at concentrations as low as 0.61 mg/L.

Effects from antimony exposure on benthic community composition have been detected at levels between 3.2 and 150 mg/kg (Long and Morgan, 1990). Data on antimony suggest an effects range-low (ER-L) of 2 mg/kg and an effects range-median (ER-M) of 25 mg/kg.

3.2 Beryllium

In environmental media, beryllium usually exists as beryllium oxide. Beryllium has limited solubility and mobility in sediment and soil.

Plants. Beryllium uptake by plants occurs when beryllium is present in the soluble form. The highest levels of beryllium are found in the roots, with lower levels in the stems and foliage (EPA, 1985a).

Soluble forms of beryllium are easily taken up by plants, probably in a manner similar to calcium and magnesium, but it is not readily translocated from roots to shoots (Peterson and Girling, 1981). Beryllium has been reported to inhibit seed germination, enzyme activation, and uptake of calcium and magnesium by roots. Common symptoms of beryllium toxicity to plants are brown, retarded roots, and stunted foliage (Romney and Childress, 1965). The phytotoxicity benchmark value for beryllium (10 mg/kg) is based on unspecified toxic effects on plants grown in surface soil amended with 10 mg/kg beryllium (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. EPA has derived an Eco-SSL for beryllium in soil based on terrestrial invertebrate toxicity. The Eco-SSL for terrestrial invertebrates (40 mg/kg) is the geometric mean of the EC_{20} values reported for pot worms (*Enchytraeus crypticus*), springtail (*Folsomia candida*), and earthworms (*Eisenia fetida*).

Mammals. The major exposure route for mammals is inhalation. Beryllium is poorly absorbed from the gastrointestinal tract, and is not absorbed through intact skin to any significant degree. Mammals exposed via inhalation exhibit pulmonary effects which may last long after exposure ceases. Based on animal studies, beryllium is poorly absorbed through both the gastrointestinal tract and the skin. The most important route of exposure for beryllium is inhalation, although absorption by this route does not appear to be extensive. Once beryllium is absorbed, it is circulated in the blood as an orthophosphate colloid and is then distributed primarily to the bone, liver, and kidneys in both humans and animals. Beryllium and its compounds are not biotransformed, but soluble beryllium compounds are partially converted to more insoluble forms in the lungs (Reeves and Vorwald, 1967).

Following inhalation of soluble beryllium compounds in both humans and animals, the lung appears to be the main target organ for toxicity. Acute exposure may cause chemical pneumonitis; chronic exposure to insoluble forms may lead to chronic beryllium disease (berylliosis), a fibrotic lung disease (Agency for Toxic Substances and Disease Registry [ATSDR], 1993a).

A variety of beryllium compounds have been demonstrated to cause pulmonary tumors following inhalation in animals. However, it is thought that oral administration does not lead to carcinogenesis due to poor absorption of the constituent from the gastrointestinal tract. The NOAEL for a rat lifetime chronic exposure to beryllium in drinking water was 0.54 mg/kg-day (Health Effects Assessment Summary Tables, 1997).

The EPA has derived an Eco-SSL for mammalian species of 21 mg/kg beryllium in soil. This mammalian Eco-SSL is the lowest calculated value based on the NOAEL for effects on growth, reproduction, and survival in mammalian herbivores (voles), mammalian ground insectivores (shrews), and mammalian carnivores (weasels).

Birds. No information was found regarding the toxicity of beryllium to birds.

Aquatic Life. Exposure routes for aquatic organisms include ingestion and gill uptake. Beryllium does not bioconcentrate in aquatic organisms. Beryllium uptake from water is low, resulting in low bioconcentration rates. Biomagnification of beryllium in aquatic food chains does not occur (Fishbein, 1981). Beryllium can be toxic to warm water fish, especially in soft water.

The Tier II secondary acute water quality value and secondary chronic water quality value for beryllium, as calculated by the method described in the EPA's *Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 35 and 0.66 micrograms per liter (μg/L), respectively.

The EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC_{20} value for beryllium is 148 μg/L. A similar value can be determined for daphnids, which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} for daphnids is 3.8 μg/L (Suter and Tsao, 1996).

3.3 Copper

Copper is ubiquitously distributed in nature in the free state and in sulfides, arsenides, chlorides, and carbonates. Several copper-containing proteins have been identified in biological systems as oxygen binding hemocyanin, cytochrome oxidase, tyrosinase, and lactase. Copper has also been identified with the development of metalloproteins employed in the sequestering and cellular detoxification of metals. Most organisms are able to regulate copper levels within their systems. Copper may accumulate in the tissues of certain organisms but it does not tend to accumulate or magnify in higher trophic levels.

Copper has been known to sorb rapidly to sediment. The rate of sorption is of course dependent upon factors such as the sediment grain size, organic fraction, pH, competing cations, and the presence of ligands. In industrialized freshwater environments around the world total copper levels within sediments can range from 7 to 2,350 ppm (Moore and Ramamoorthy, 1984).

Plants. Copper is an essential nutrient for the growth of plants. Background concentrations of copper in grasses and clovers collected in the United States averaged 9.6 mg/kg and 16.2 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Copper is one of the least mobile heavy metals in soil, and its availability to plants is highly dependent on the molecular weight of soluble copper complexes (Kabata-Pendias and Pendias, 1992).

According to Rhodes et al. (1989), copper concentrations in plant tissues do not serve as conclusive evidence of copper toxicity in species of plants such as tomatoes, because some species are able to tolerate higher concentrations of copper than others. The pH of soil may also influence the availability and toxicity of copper in soils to plants. In a study with tomato plants, Rhodes et al. (1989) found a reduction in plant growth when plants were grown in soils containing greater than 150 mg/kg of copper at a pH of less than 6.5. At pH values greater than 6.5, soil copper concentrations of greater than 330 mg/kg were required to reduce plant growth.

Concentrations of copper in leaf tissue that are excessive or toxic to various plant species range from 20 to 100 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 100 mg/kg has been proposed by Efroymson et al., (1997) as a benchmark screening value for copper phytotoxicity in soil. General symptoms of copper toxicity in plants include the presence of dark green leaves followed by induced iron chlorosis; thick, short, or barbed wire roots; and depressed tillering (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Beyer et al. (1982) and others have reported that copper concentrations in earthworms have been observed to be correlated with copper concentrations in soil. Further studies by Beyer (1990) indicate that copper can be more toxic to bioturbative earthworms than most metals. Research by Phillips (1980) suggests that copper and other metal accumulation within terrestrial invertebrates may vary significantly depending on soil conditions and other physical/chemical properties, and bioconcentration factors can approach 10,000. EPA (2000a) has derived a soil screening level (SSL) for copper of 61 mg/kg. This invertebrate SSL was based on reproductive and growth data from studies conducted with natural soils under conditions of high or very high bioavailability. The tests were conducted with highly soluble salts and neither aging nor weathering, which would lower bioavailability, was included in the experimental designs.

Mammals. Copper is an essential trace element to animals as well as plants (Callahan et al., 1979), but becomes toxic at concentrations only slightly higher than essential levels (EPA, 1985b). Copper is an essential element for hemoglobin synthesis and oxidative enzymes in animals, and is absorbed by mammals following ingestion, inhalation, and dermal exposure. Once absorbed, copper is distributed to the liver, and is not metabolized (Marceau et al., 1970). No evidence of bioaccumulation was obtained in a study of pollutant concentrations in the muscles and livers of 10 species of herbivorous, omnivorous, and carnivorous animals in Donana National Park in Spain (Hernandez et al., 1985). Copper concentrations in small mammals collected from various uncontaminated sites ranged from 8.3 to 13.4 mg/kg (whole-body concentrations) (Talmage and Walton, 1991). Highest concentrations of copper tend to be in hair, followed in decreasing concentration by liver, kidney, and whole body (Hunter and Johnson, 1982). Among the small mammals collected, Hunter and Johnson (1982) found shrews (*Sorex araneus*) to contain the highest concentrations of copper. Mice were found to contain the lowest copper concentrations. Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg-day of copper as copper sulfate (Lecyk, 1980). Increased mortality rates in mink offspring have been observed at levels above 3.21 mg/kg-day (Aulerich et al., 1982).

Laboratory toxicity data for mink exposed to copper sulfate in their diet were used to estimate a NOAEL value of 11.7 mg/kg/day (Aulerich et al., 1982). Reproduction was the endpoint studied. Symptoms of acute copper poisoning in mammals include vomiting, hypotension, melena, coma, jaundice, and death (Klaassen et al., 1991). Selenium can act as an antidote for copper poisoning.

Birds. Laboratory toxicity data for one-day old chicks exposed to copper oxide in their diets were used to estimate a NOAEL value of 47 mg/kg/day (Mehring et al., 1960). Growth and mortality were the endpoints studied.

Aquatic Life. Invertebrates inhabiting "polluted" freshwaters worldwide have been known to have tissue residues of copper ranging from 5 to 200 ppm (Moore and Ramamoorthy, 1984). Field studies have shown that there is virtually no accumulation of this metal through the food chain (Fuller and Averett, 1975). Studies by Kosalwat and Knight (1987) indicated that copper present in the substrate or sediment was significantly less toxic to chironomid species than overlying water column levels. The substrate copper concentration at which chironomid larval growth was reduced 50 percent (EC_{50}) was 1,602 mg/kg. These researchers found that deformities in larval mouth parts were observed in elevated concentrations, and adult emergence was inhibited when the sediment concentration exceeded 1,800 mg/kg. Carins et al. (1984) reported copper toxicity in sediment for several chironomus midges and cladocerans with LC_{50} values ranging from 681 to 2,296 mg/kg.

Moore and Ramamoorthy (1984) reported that copper can be highly toxic to many aquatic plants and algae. Inhibition of growth can occur at levels as low as 0.1 mg/L. In some algal species, copper may inhibit electron transport during photosynthesis. In general, since low pH increases the proportion of free ions in solution, acidic waters may exhibit greater copper toxicity. However, Stokes (1975) reported the observance of algal adaptation to copper-tainted waters with certain species able to tolerate and flourish within highly copper-contaminated waters.

Moore and Ramamoorthy (1984) reported LC_{50} in fresh water fish ranging from 0.017 to 1.0 mg/L. Copper is similar to other metals in that its toxicity to fin fish is often greater within fresh water environments versus marine environments because of the lack of complexing agents within fresh water.

3.4 Lead

Global production of lead from both smelter and mining operations has been high throughout the past century. Lead is commonly used in storage batteries as well as in ammunition, solder, and casting materials. In addition, tetraethyl lead was a principal additive to gasoline as an anti-knock agent and was commonly used as an additive in paints. In short, lead is one of the most ubiquitous pollutants in the civilized world.

Lead is strongly sorbed in sediments, and the rate is strongly correlated with grain size and organic content. In the absence of soluble complexing species, lead is almost totally adsorbed to clay particles at pH values greater than 6 (Moore and Ramamoorthy, 1984).

Plants. Although lead is not an essential nutrient for plant growth, it is detected in plant tissues due to the prevalence of lead in the environment. The bioavailability to plants of lead in soil is limited. Bioavailability may be enhanced by a reduction in soil pH, a reduction in the content of organic matter and inorganic colloids in soil, a reduction in iron oxide and phosphorous content, and increased amounts of lead in soil (National Research Council of Canada [NRCC], 1973). Plants can absorb lead from soil and air. Aerial deposition of lead can also contribute significantly to the concentration of lead in above-ground plant parts. Lead is believed to be the metal of least bioavailability and the most highly accumulated metal in root tissue (Kabata-Pendias and Pendias, 1992). Lead tends not to accumulate into plants from soil unless concentrations are very high (i.e., percentage levels). The tips of some trees, such as pine and fir, can accumulate lead from contaminated soil when contamination levels are high. Such conditions often occur at mining sites (National Library of Medicine, 1996). Lead inhibits plant growth, reduces photosynthesis, and reduces mitosis and water absorption

Mean background concentrations of lead in grasses and clovers have been reported to range from 2.1 to 2.5 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Adverse effects of lead on terrestrial plants occur only at total concentrations of several hundred mg/kg of soil (Eisler, 1988). This is explained by the fact that, in most cases, lead is tightly bound to soils, and substantial amounts must accumulate before it can affect the growth of higher plants (Boggess, 1977).

The Eco-SSL for plants, as derived by EPA (2005b), is 110 mg/kg. The plant Eco-SSL is the geometric mean of the maximum acceptable toxicant concentration (MATC) for four test species (loblolly pine, red maple, Berseem clover, and ryegrass) under three different test conditions.

Terrestrial Invertebrates. Lead has been shown to accumulate in the tissues of lower trophic level organisms, including terrestrial invertebrates, but is not effectively transferred to higher trophic level organisms through the food web. Centipedes (*Lithobius variegatus*) that ate woodlice hepatopancreas did not assimilate lead even though the food contained concentrations that were many times greater than normally encountered. However, survival and reproduction were reduced in woodlice (*Porcellio scaber*) fed soil litter treated with 12,800 mg/kg lead (Beyer and Anderson, 1985). This concentration of lead is similar to the amount of lead reportedly associated with reductions in natural populations of decomposers, such as fungi, earthworms, and arthropods.

EPA (2005b) has derived an Eco-SSL based on soil invertebrate toxicity of 1,700 mg/kg lead in soil. The Eco-SSL for terrestrial invertebrates is the geometric mean of the MATC values for one test species (*Folsomia candida*) under three different test conditions.

Mammals. As with plants, lead is not considered an essential nutrient for mammalian life. Ingestion is the major route of exposure for wildlife. Lead tends to accumulate in bone, hair, and teeth. Biomagnification of lead is negligible (Eisler, 1988). Jenkins (1981) also reported that soil conditions of low alkalinity and low pH can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower animals, and higher plants. Reduced survival was reported at acute oral doses as low as 5 mg/kg body weight in rats, at a chronic dose of 0.3 mg/kg body weight in dogs, and at a dietary level of 1.7 mg/kg body weight in horses (Eisler, 1988). Laboratory data from studies of rats fed lead acetate in their diets were used to estimate a NOAEL value of 8.0 mg/kg-day (Azar et al., 1973). Reproduction was the endpoint for this study. Symptoms of lead poisoning in mammals are diverse and depend on the form of lead ingested, the concentration, and the species and its age. These symptoms may include reproductive impairment, decreased body weight, vomiting, uncoordinated body movements, visual impairment, reduced life span, renal disorders, and abnormal social behavior (Eisler, 1988).

In laboratory studies, breeding mice exposed to low doses of lead in drinking water (25 ppm) resulted in loss of the strain in two generations with many abnormalities (Schroeder and Mitchener, 1971). Exposure of rats in this same experiment resulted in many early deaths and runts. Blood δ-aminolevulinic acid dehydratase activity associated with exposure to lead was reduced in white-footed mice living near a metal smelter (Beyer et al., 1985). Amounts of whole-body lead content and feeding habits of roadside rodents have been correlated with highest body burdens in insectivores such as shrews, intermediate in herbivores, and lowest in granivores (Boggess, 1977; Getz et al., 1977).

EPA (2005b) has derived an Eco-SSL for lead in soil of 56 mg/kg for the protection of mammalian species. This mammalian Eco-SSL for lead is based on the NOAEL for reproduction, growth and survival in a number of mammalian species.

Birds. Most of the information on the effects of lead to terrestrial vertebrates is concerned with acute poisoning of waterfowl by lead shot. Apparent symptoms include loss of appetite and mobility, avoidance of other birds, lethargy, weakness, emaciation, tremors, dropped wings, green feces, impaired locomotion, loss of balance and depth perception, nervous system damage, inhibition of heme synthesis, damage to kidneys and liver, and death (Eisler, 1988; Mudge, 1983). Anemia, kidney disease, testicular and liver lesions, and neurological disorders have been associated with high brain lead concentrations in mourning doves (*Zenaida macroura*) (Kendall, 1992). Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively more tolerant to moderate lead exposure, including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2,000 ppm (Hoffman et al., 1985).

Toxicity of lead to birds is dependent upon the form of lead, the route of exposure and exposure duration, and the species and age of the bird. Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively tolerant to moderate lead exposure (Eisler, 1988). Laboratory toxicity data for American kestrels fed metallic lead in their diet were used to estimate a NOAEL value of 3.85 mg/kg-day (Pattee, 1984). Reproduction was the endpoint for this study.

An avian Eco-SSL for lead has been derived by EPA (2005b) to be 11 mg/kg lead in soil. This avian Eco-SSL for lead is based on the NOAEL for reproduction, growth and survival in a number of avian species.

Aquatic Life. All life stages are sensitive to the toxic effects of lead; however, embryos are more sensitive to lead than are later juvenile stages (Davies et al., 1976). Lead uptake depends on exposure time, aqueous concentration, pH, temperature, salinity, diet, and other factors. For example, gill, liver, kidney, and erythrocytes accumulate lead from aqueous sources in proportion to exposure time and concentration (Holcombe et al., 1976). Direct erythrocyte injury is considered the first and most important sign of lead poisoning in catfish (Dawson, 1935). Respiratory distress occurs in fish living in rivers receiving lead mining wastes in England

(Carpenter, 1924; 1925; 1926). Fish are thought to be asphyxiated as a result of a mucous coating over the gills (National Academy of Sciences [NAS], 1972).

No significant biomagnification of lead occurs in aquatic ecosystems (Boggess, 1977). Background concentrations of lead in fish tend to be less than 1 mg/kg (dry weight) (Eisler, 1988). The EPA's National Recommended Water Quality Criteria for lead in freshwater are 65 µg/L for acute exposure and 2.5 µg/L for chronic exposure (EPA, 1999b). In general, dissolved lead is more toxic than total lead, and organic forms of lead are more toxic than inorganic forms. Soluble lead in the water column becomes less bioavailable as water hardness increases. Chronic exposure of fish to lead may result in signs of lead poisoning such as spinal curvature, anemia, darkening of the dorsal tail region, destruction of spinal neurons, difficulties in swimming, growth inhibition, changes in blood chemistry, retarded sexual development, and death (Eisler, 1988).

Physicochemical conditions within the water may also affect lead uptake and toxicity. Under conditions of low alkalinity (less than 50 microequivalents per liter) and low pH, lead can accumulate in fish, algae, mollusks, and benthic invertebrates (Wiener and Stokes, 1990). Irwin (1988) reported significant accumulations of lead in the Trinity River within mosquitofish, turtles, bullhead minnows, and crayfish. Nevertheless, lead concentrations were not higher in top-of-the-food-chain predators like gar than they were in mosquitofish, suggesting minimal biomagnification of lead.

The majority of benthic invertebrates do not bioconcentrate lead from water or abiotic sediment particles. There is some evidence of bioaccumulation through the food web of organic forms of lead, such as tetraethyl lead. Anderson et al. (1980) reported a lead LC_{50} of 258 ppm for the chironomid and that growth of this organism was not reduced above this level in freshwater sediments. In addition, Suter and Tsao (1996) reported effect levels in the water flea (*Daphnia magna*) to be in the 12.26 parts per billion (ppb) range, while Khangrot and Ray (1989) reported an LC_{50} of 4.89 ppm for *D. magna*.

3.5 Zinc

Zinc is a naturally occurring element that may be found in both organic and inorganic forms and, as such, is commonly found in the environment. In general, zinc is concentrated in the sediments of water bodies. The NAS (1979) has reported that zinc will probably be detected in 75 percent of all water bodies examined for the compound at various locations. The fate of zinc in soils appears to have a pH basis. Studies have shown that a pH of less than 7 often favors zinc desorption (EPA, 1984).

Plants. Background concentrations of zinc in terrestrial plants range from 25 to 150 mg/kg (dry weight) (NAS, 1979). The deficiency content of zinc in plants is between 10 and 20 ppm (dry weight). Roots often contain the highest concentrations of zinc (Kabata-Pendias and Pendias, 1992).

Certain species of plants, particularly those from the families Caryophyllaceae, Cyperaceae, and Plumbaginaceae, and some tree species are extremely tolerant to elevated zinc concentrations (Kabata-Pendias and Pendias, 1992). Concentrations of zinc in these plants may reach 1 percent (dry weight) in the plant. Concentrations in leaf tissue that are excessive or toxic to various plant species range from 100 to 400 mg/kg. Concentrations of 100 to 500 mg/kg are expected to result in a 10 percent loss in crop yield (Kabata-Pendias and Pendias, 1992). General symptoms of zinc toxicity in plants include the presence of chlorotic and necrotic leaf tips, interveinal chlorosis in new leaves, retarded growth of the entire plant, and injured roots that resemble barbed wire (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. EPA (2000a) has developed an ecological soil screening level (SSL) for zinc in soil of 120 mg/kg. This SSL was based on reproduction and population effects in experiments conducted with natural soils under conditions of high or very high zinc bioavailability. It is also important to note that in studies conducted with mixtures of cadmium, copper, and zinc, it was concluded that the three metals acted antagonistically. It has also been shown that a decrease in pH and/or organic matter in the soil tends to decrease the concentration of zinc in soil at which toxic effects are observed (Spurgeon and Hopkin, 1996). Zinc has been shown to accumulate in earthworm species (Beyer et al., 1982) but generally is not biomagnified through the food web.

Mammals. Zinc is an essential trace element for normal fetal growth and development. However, exposure to high levels of zinc in the diet has been associated with reduced fetal weights, altered concentrations of fetal iron and copper, and reduced growth in offspring (Cox et al., 1969). Poisoning has been observed in ferrets and mink from chewing corroded galvanized cages (Clark et al., 1981). Symptoms of zinc toxicity include lassitude, slower tendon reflexes, bloody enteritis, diarrhea, lowered leukocyte count, depression of the central nervous system, and paralysis of the extremities (Venugopal and Luckey, 1978). A study by Kinnamon (1963) showed a NOAEL for oral exposure to a zinc compound over a period of 73 days to be 250 mg/kg body weight, and mice given 500 mg/L of zinc as zinc sulfate in drinking-water have shown hypertrophy of the adrenal cortex and pancreas. Young animals are much more susceptible to poisoning by zinc than are mature animals (Clark et al., 1981).

Animals are quite tolerant of high concentrations of zinc in the diet. Levels 100 times that required in the diet usually do not cause detectable symptoms of toxicosis (NAS, 1979). Laboratory data for rats exposed to zinc oxide in their diet were used to estimate a NOAEL value of 160 mg/kg-day (Schlicker and Cox, 1968). Reproduction was the endpoint studied. Symptoms of zinc poisoning in mammals include lameness, acute diarrhea, and vomiting (Eisler, 1993).

Birds. Dietary zinc concentrations of greater than 2,000 mg/kg are known to result in reduced growth of domestic poultry and wild birds (Eisler, 1993). Reduced survival has been documented at zinc concentrations greater than 3,000 mg/kg diet or at a single dose of greater than 742 mg/kg body weight (Eisler, 1993). Laboratory data for white leghorn hens exposed to zinc sulfate in their diet were used to estimate a NOAEL value of 14.5 mg/kg-day (Stahl et al., 1990). Reproduction was the endpoint for this study. A value of 51 mg/L has been calculated as the NOAEL for chronic exposure of birds to zinc carbonate in drinking water (Sample et al., 1996).

Aquatic Life. Zinc residues in freshwater and marine fish are generally much lower than those found in algae and invertebrates. Thus, there is little evidence for bioaccumulation (Moore and Ramamoorthy, 1984). Rainbow trout (*Oncorhyncus mykiss*) have the ability to detect and avoid areas of water containing 5.6 ppb zinc (Sprague, 1968). Cairns and Scheier (1968) reported 96-hour LC_{50}s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC_{50}s ranging from 2.86 to 3.78 ppm in soft water. These results demonstrate that water hardness affects the toxicity of zinc to fish. Chronic toxicity tests have been conducted with five species of freshwater fish. Chronic values ranged from 47 µg/L for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinallis*) (EPA, 1980a).

Acute toxicity to freshwater invertebrates is relatively low and, as with other metals, increasing water hardness decreases the toxicity of zinc (Moore and Ramamoorthy, 1984). As reported by Baudouin and Scoppa (1974), the 48-hour LC_{50} for the cladaceran *Daphnia hyalina* was 0.055 mg/L, and 5.5 mg/L for the copepod *Cyclops abyssorum*. Four chronic toxicity tests are reported for *Daphnia magna*, with chronic values ranging from 47 µg/L to 136 µg/L (EPA, 1980a). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1980a).

3.6 Gamma-Chlordane

Chlordane is a chlorinated pesticide whose commercial use has been banned in the United States since 1988 (EPA, 1988). Technical chlordane is a mixture of some 50 different compounds; the major constituents being alpha-, beta, and gamma-chlordane (Howard, 1991). Chlordane is generally immobile in soil; however, movement to groundwater can occur (Howard, 1991). The compound is biodegraded at a very slow rate (Howard, 1991). In aquatic systems, chlordane is not expected to undergo significant hydrolysis, oxidation, or direct photolysis (Howard, 1991). Because of its widespread use in the past and its low biodegradation rate, chlordane is commonly measured in low concentrations in environmental samples (Eisler, 1990). Chlordane, with an organic carbon-sediment partition coefficient (K_{oc}) of between 15,500 and 24,600, strongly adsorbs to sediment (Howard, 1991).

Plants. Information on the toxicity of chlordane in plants is very limited. Chlordane concentrations greater than 100 µg/L were found to inhibit the growth of blue-green algae (Glooschenko et al., 1979), whereas concentrations greater than 1,000 µg/L inhibited the growth of green algae (Glooschenko and Lott, 1977). A 1971 nationwide survey of chlordane concentrations in corn (*Zea mays*) and sorghum (*Sorgum halepense*) indicated 0.48 mg/kg (dry weight) in corn kernels, 1.26 mg/kg (dry weight) in corn stalk, and 0.42 mg/kg (dry weight) in sorghum (Carey et al., 1978).

Terrestrial Invertebrates. No information was found regarding the potential toxicity of chlordane to terrestrial invertebrates.

Mammals. Absorption of chlordane can occur through the skin, diet, and inhalation (Eisler, 1990). Studies have shown the metabolites of technical grade chlordane to be present in bat milk (Clark et al., 1978). Oxychlordane and heptachlor epoxide are metabolites of the components of technical grade chlordane (Eisler, 1990). Oxychlordane is more toxic and persistent than any of the parent compounds.

Acute oral LD_{50} values for exposure of sensitive mammalian species to technical grade chlordane range from 25 to 50 mg/kg body weight. Chlordane fed to rats at 2.5 mg/kg caused slight liver damage (National Research Council [NRC], 1977). Growth retardation and liver damage were observed in rats fed 150 and 300 mg/kg chlordane in their diets over a two-year period (Clayton and Clayton, 1982). No measurable effects were found in Cynomolgus monkeys (*Macaca spp.*) exposed to technical grade chlordane concentrations of 10 µg/L in air for 90 days (Khasawinah et al., 1989). Based on a study of mice fed technical grade chlordane in their diets, a NOAEL value of 4.6 mg/kg/day has been derived (Keplinger et al., 1968). The critical endpoint for this study was reproduction.

Symptoms of acute chlordane poisoning in mammals include diarrhea, avoidance of food and water, hair loss, hunched appearance, abdominal distension, labored respiration, salivation, muscle tremors, incoordination, convulsions, and death in some cases (Eisler, 1990). Chlordane has been reported to be carcinogenic in mice (Eisler, 1990) and teratogenic in rats (NRC, 1977). Chlordane potentiates aldrin, endrin, and parathion toxicity in mice (Jones et al., 1977).

Birds. Chlordane and related metabolites are commonly detected in wild birds throughout the United States. A 1982 survey revealed 45 percent of the captured European starlings (*Sturnus vulgaris*) contained detectable concentrations of oxychlordane (Eisler, 1990). Avian studies have shown older birds and raptors to contain relatively high concentrations of chlordane and its metabolites (Eisler, 1990). Lethal concentrations of oxychlordane and heptachlor have been measured in raptors that consumed either poisoned bait or prey (Blus et al., 1983). As in mammals, chlordane concentrations are greatest in tissues high in lipid content (Eisler, 1990). Biological half-lives of chlordane isomers in birds range from 11.2 to 35.4 years (Eisler, 1990).

Among the birds tested, California quail (*Callipepla californica*) are the most sensitive to chlordane, with an acute oral LD_{50} of 14.1 mg/kg body weight (Hudson et al., 1984). Toxic responses to technical chlordane are primarily attributed to the metabolite oxychlordane (Stickel et al., 1983). Oxychlordane concentrations in brain tissue greater than 5 mg/kg (wet weight) are considered lethal to birds (Stickel et al., 1979). Based on exposure of red-winged blackbirds to chlordane in their diets for 84 days, a chronic NOAEL value of 2.14 mg/kg/day has been derived (Stickel et al., 1983). The critical endpoint in this study was mortality. Signs of chlordane poisoning in birds include sluggishness, drooped eyelids, fluffed feathers, altered resting patterns, reduced appetite, weight loss, quivering and panting, neck arched over back, and arched back (Stickel et al., 1983).

Aquatic Life. Because of the widespread historical use of chlordane, it is commonly detected in aquatic environments. Vieth et al. (1979) reported 36 percent of all fish samples collected in domestic watershed throughout the United States in 1976 to contain chlordane. Of the components of technical grade chlordane detected in fish, cis-chlordane was the most abundant, followed by trans-nonachlor, trans-chlordane, and cis-nonachlor (Ribick and Zajicek, 1983). Bioconcentration factors for aquatic organisms range from 108 for frogs (*Xenopus laevis*) to 38,000 for fathead minnows (*Pimephales promelas*) (Howard, 1991). There is some evidence of biomagnification of chlordane in freshwater fish (Eisler, 1990).

Variations in the response of aquatic organisms to chlordane are related to both biological factors of the biota and physical factors of chlordane and the aquatic environment. Younger life-stages are more sensitive to chlordane toxicity than older life-stages (Eisler, 1990). In addition, the health and lipid content of the organism can have an impact on the response (Eisler, 1990). Water temperature, salinity, and sediment loading are abiotic factors that influence the bioavailability of chlordane (NRCC, 1975). Cis-chlordane appears to be more toxic than trans-chlordane, and the photoisomers seem to be more toxic than the parent compounds (Eisler, 1990).

The EPA's National Ambient Water Quality Criteria for chlordane is 2.4 μg/L for acute exposure and 0.0043 μg/L for chronic exposure of aquatic life (EPA, 2003). The water quality criteria for chlordane represent the sum of alpha-chlordane, gamma-chlordane, alpha-chlordene, gamma-chlordene, alpha-nonachlor, gamma-nonachlor, and oxy-chlordane. The lowest chronic toxicity values for chlordane reported in the literature for fish and daphnids are 1.6 and 16 μg/L, respectively (Suter and Tsao, 1996).

The test EC_{20} for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of chlordane causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The fish EC_{20} value for fish is less than 0.25 μg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration of chlordane causing less than 20 percent reduction in the growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC_{20} benchmark for daphnids is 12.1 μg/L (Suter and Tsao, 1996). Signs of chlordane poisoning in fish include hyperexcitability, increased respiration rate, erratic swimming, loss of equilibrium, convulsions, and death (NRCC, 1975).

4.0 Fate and Transport

The environmental fate and transport of the COPECs in the various media at the BBGR Ranges will govern the potential for exposures to wildlife. In general, COPECs in environmental media may be available for direct exposure (e.g., plants exposed to surface soil) and they may also have the potential to migrate to other environmental media or areas of the site. This chapter addresses the mechanisms by which COPECs can be transported and the chemical properties that determine their transport.

4.1 Fate and Transport in Soil

Contaminants in surface soil at the BBGR Ranges have the potential to be transported from their source area to other areas within the respective ranges and to off-site locations by a number of mechanisms, including volatilization, dust entrainment, surface runoff, and infiltration to subsurface soil/groundwater.

Several volatile organic compounds (VOC) were identified in the upper soil horizons at the BBGR Ranges. These VOCs have a high potential to volatilize to the atmosphere and be transported from their source area via air movement. The concentrations of VOCs detected in surface soil at the BBGR Ranges are low; therefore, this transport mechanism is expected to be insignificant with respect to other transport mechanisms active at this site. Most of the metals and SVOCs in the surface soil at the BBGR Ranges are not expected to volatilize to any great extent, with the exception of mercury, which would be expected to volatilize relatively rapidly. Most of the metals and SVOCs in the surface soil at the BBGR Ranges are generally closely associated with particulate matter and would be transported from their source areas by fugitive dust generation and entrainment by the wind. Subsequent dispersion by atmospheric mixing could transport particulate-associated contaminants to other parts of the BBGR Ranges and to off-site locations. The generation of fugitive dust and subsequent transport by the wind is potentially a significant transport mechanism at the BBGR Ranges, based on the presence of non-vegetated areas and areas of sparse vegetation within certain areas of these ranges (e.g., impact areas and soil berms).

The transport of surface soil-associated contaminants by surface runoff is another potentially significant transport mechanism. Surface soil contaminants may be solubilized by rainwater and subsequently transported to drainage ditches, low-lying areas, and tributaries of South Branch of Cane Creek and Ingram Creek via surface runoff. The solubility of inorganics in rainwater is largely dependent upon the pH of the rainwater. Because the rainwater in this region is most likely slightly acidic, the inorganic constituents in surface soil are likely to solubilize to some degree in the rainwater and be subject to transport via runoff. Most of the semivolatile compounds are strongly associated with soil particles and would not solubilize to a large extent. Contaminants that may be more strongly bound to particulate matter in surface soil (e.g., SVOCs and some of the inorganics) may be entrained in surface water runoff and transported to drainage ditches, low-lying areas, and tributaries of South Branch of Creek and Ingram Creek via surface runoff. Many of the metals and SVOCs are strongly sorbed to soil particles and could be transported from their source areas via this mechanism.

Contaminants in surface soil may be transported vertically to subsurface soils and groundwater via solubilization in rainwater and infiltration. Subsequent groundwater transport to surface water in the South Branch of Cane Creek and Ingram Creek tributaries could result in exposure of aquatic receptors to soil contaminants. Migration in this manner is dependent upon contaminant solubility and frequency of rainfall. The soil type (rough, stony land) in the vicinity of the BBGR Ranges does not promote rapid infiltration, but rather is more conducive to the promotion of surface runoff. Based on the constituents detected in surface soil and the soil type found at these sites, vertical migration of surface soil constituents is expected to be minimal at the BBGR Ranges. Surface water and groundwater monitoring data support the theory that the vertical transport of constituents is relatively insignificant at the BBGR Ranges as no constituents were detected at elevated concentrations in surface water and several pesticides and nitroaromatic compounds were only sporadically detected at low levels in groundwater.

The transfer of contaminants in surface soil to terrestrial plants through root uptake and transfer to terrestrial animals through ingestion and other pathways are potentially significant transfer mechanisms. Many metals are readily absorbed from soil by plants, but they are not biomagnified to a great extent through the food web. There are several exceptions to this, namely, arsenic and nickel, which may bioconcentrate and/or biomagnify (ATSDR, 1989; 1995). Many of the SVOCs have the potential to bioaccumulate in lower trophic level organisms (e.g., terrestrial invertebrates), but most higher trophic level animals have the ability to metabolize these compounds rapidly, precluding the potential for bioconcentration (Eisler, 1987).

VOCs in the surface soil at the BBGR Ranges are expected to volatilize and/or photolyze rapidly (half-lives of 3 hours to 5 days) when exposed to sunlight (Burrows et al., 1989). The other surface soil contaminants (metals and SVOCs) are expected to remain in the soil relatively unchanged by physical and/or chemical processes for much longer periods of time.

Chlorinated pesticides and herbicides detected in surface soil at the BBGR Ranges could bioaccumulate significantly and could also biomagnify in the food web. However, these constituents were detected sporadically and at relatively low concentrations.

4.2 Fate and Transport in Surface Water

It is important to preface any discussion about surface water transport with the fact that no constituents were detected is surface water samples from the BBGR Ranges at elevated concentrations. Constituents in surface water at the BBGR Ranges may be transported from their sources to other areas at the ranges or to off-site locations by the following mechanisms: 1) volatilization, 2) transfer to groundwater, 3) transfer to sediment, and 4) flow downstream. In general, the hills east and south of the BBGR Ranges contain the headwaters of South Branch of Cane Creek and Ingram Creek. Water in South Branch of Cane Creek and Ingram Creek originates mainly from overland flow from the surrounding watershed and from seeps located in the surrounding mountains. There also appears to be sporadic and localized contributions to creek flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution in South Branch of Cane Creek and Ingram Creek from groundwater varies according to the amount of precipitation, with an increase of groundwater contribution when precipitation raises the potentiometric surface.

Thus, constituents in groundwater could migrate to surface water in South Branch of Cane Creek and Ingram Creek. This transport mechanism appears to be relatively insignificant based on the fact that no constituents were detected at elevated concentrations in surface water. Constituent transfer to sediments represents another significant transfer mechanism, especially where constituents are in the form of suspended solids, or are hydrophobic substances (e.g., PAHs) that can become adsorbed to organic matter in the sediments. The metals detected in surface water have the potential to associate with suspended particulate matter. VOCs in surface water would be expected to rapidly volatilize from the water-air interface and be dispersed in the atmosphere. Therefore, transport of VOCs in surface water is not expected to occur for any significant distance.

Constituents in surface water can be transported to other ranges within the BBGR range complex or off-site via South Branch of Cane Creek or Ingram Creek. Transfer of constituents in surface water to aquatic organisms is also a potentially significant transfer pathway. Some of the inorganic constituents detected in surface water may bioaccumulate in lower trophic level organisms. Most of the inorganics detected in surface water are not highly bioconcentratable; therefore, transfer through the food web is expected to be minimal for these compounds.

4.3 Fate and Transport in Sediment

Constituent transfer between sediment and surface water potentially represents a significant transfer mechanism, especially when contaminants are in the form of suspended solids. Sediment/surface water transfer is reversible; sediments often act as temporary repositories for constituents and gradually release constituents to surface waters. This is especially true in surface water systems that are acidic, as is the case with South Branch of Cane Creek and Ingram Creek in the vicinity of the BBGR Ranges. Sorbed or settled contaminants can be transported with the sediment to downstream locations. Much of the substrate of South Branch of Cane Creek and Ingram Creek and their tributaries in the vicinity of the BBGR Ranges is best characterized as gravel or cobbles. Very few areas of high organic content sediment or muck are present. The very low organic content of gravel and cobble create a substrate with very low binding capacity; therefore, constituents released to South Branch of Cane Creek or Ingram Creek and their tributaries via surface runoff or other transport mechanisms would most likely remain suspended in the surface water, be transported downstream, and would not be sequestered in the stream substrate directly adjacent to the BBGR Ranges.

Although transfer of sediment-associated constituents to bottom-dwelling biota also represents a potentially significant transfer mechanism, it is not expected to be a major mechanism at the BBGR Ranges. Lower trophic level organisms may accumulate metals and PAHs; however, higher trophic level organisms have the ability to metabolize PAHs and therefore reduce their accumulative properties. Most of the inorganics detected in sediment are not bioaccumulative. Mercury and copper may bioaccumulate to some extent due to exposures to sediment.

4.4 Constituent-Specific Fate and Transport Properties

The following sub-sections describe the fate and transport properties of each of the COPECs identified at the BBGR Ranges.

4.4.1 Antimony

Little is known of the adsorptive behavior of antimony, its compounds, and ions. The binding of antimony to soil is determined by the nature of the soil and the form of antimony deposited on the soil. Some forms of antimony may bind to inorganic ligands. On the other hand, a mineral form would be unavailable for binding. Since antimony has an anionic character, it is expected to have little affinity for organic carbon (ATSDR, 1992). Antimony binds to soil, particularly to particles containing iron, manganese, or aluminum (ATSDR, 1992). Bodek *et al.* (1988) indicate that antimony oxides are highly soluble, which suggests environmental mobility. However, Callahan *et al.* (1979) indicate that antimony may have an affinity for clay and other mineral surfaces. Therefore, it can be concluded that the fate of antimony in soil is somewhat uncertain, and dependent upon many inter-related environmental factors. Antimony is also oxidized by bacteria in the soil. In water, antimony is oxidized when exposed to atmospheric oxygen.

Antimony does not appear to bioconcentrate appreciably in fish and aquatic organisms. No detectable bioconcentration occurred during a 28-day test using bluegills (USEPA, 1980). Bioconcentration factors for antimony ranged from 0.15 to 390 (Callahan, 1978). Uptake of antimony from soil by plants is minor and appears to be correlated with the amount of antimony that is soluble (Ainsworth, 1988). Antimony is not significantly metabolized and is excreted in the urine and feces. It does not biomagnify in terrestrial food chains, but can bioconcentrate to a slight degree in aquatic organisms. Antimony bioconcentration was measured in voles, shrews, rabbits, and invertebrates around a smelter. Analysis of antimony in organs of the small mammals, compared with estimates of their antimony intake from food, showed that, although the amount of antimony in the organs was elevated, it was low compared to the amount ingested. The results suggest that antimony does not biomagnify from lower to higher trophic levels in the food chain (ATSDR, 1992). It should also be noted that antimony is associated with ammunition, being present in lead alloys in bullets and in materials used as primers. Antimony can be present in both the +3 and +5 valence states, depending on pH, oxidation-reduction potential, and several other chemical properties of the environmental medium in which it is found. Antimony can methylate via chemical and/or biological reactions into an organic form under reducing conditions such as those commonly found within highly organic fine sediments and hydric soils (ATSDR, 1992).

4.4.2 Beryllium

Beryllium is naturally present in soil and sediment. Beryllium in air is attached to particulate matter whose residence time in air is dependent upon particle size. The transport of beryllium from the atmosphere to terrestrial and aquatic surfaces occurs through wet and dry deposition (ATSDR, 2002).

Beryllium binds strongly to soil fulvic acid, with binding increasing with increasing pH. However, beryllium has a much stronger affinity for clay minerals than for organic matter. Beryllium is usually associated in soil with aluminum sites on clay minerals rather than with iron oxides. In highly alkaline soils, the mobility of beryllium may increase as a result of the formation of soluble hydroxide complexes (Callahan, et al., 1979). In acidic soils, dissolved Be^{2+} has been found to be the prevailing beryllium species in the soil solution, and it should be relatively mobile in these environments (Kram, et al., 1998). The sediment-water distribution coefficients (K_d) for beryllium are very high, indicating a very low mobility in sediment

(ATSDR, 2002). In sediments from Lake Michigan, K_d values ranged between 105 and 106 (Hawley, et al., 1986).

The concentration of beryllium in plants is very low. Soluble forms of beryllium must be present for uptake to occur in plants. The vast majority of beryllium in plant tissues is found in the roots; very little is translocated to the foliage or fruits. Beryllium does not bioconcentrate in aquatic organisms (ATSDR, 2002). A measured bioconcentration factor (BCF) of 19 was reported for beryllium in bluegill (EPA, 1980b). Other investigators have reported a BCF of 100 for freshwater and marine plants, vertebrates, and fish (Callahan et al., 1979). Very low bioaccumulation of beryllium was observed in southern toads (*Bufo terrestris*) exposed directly to elevated levels of beryllium (Hopkins et al., 1998).

Beryllium exhibits only the +2 oxidation state in water. In the pH range of 6-8, typical of most waters, the speciation of beryllium is controlled by the formation of solid beryllium hydroxide, $Be(OH)_2$, which has a very low solubility. Therefore, most beryllium in natural waters is found in association with particulates rather than in the dissolved form (Callahan, et al., 1979). Other transformations of environmental importance are the formation of insoluble basic carbonates, such as $(BeCO_3)_2Be(OH)_2$, formed by reaction of dissolved carbonate with beryllium solutions and the formation of beryllium sulfate (i.e., $BeSO_4$) formed by reaction of soluble sulfates with beryllium solutions. Typical transformation processes for beryllium in soil include precipitation, complexation, and anion exchange (ATSDR, 2002).

4.4.3 Copper

In general, adsorption is probably the most important controlling mechanism in determining copper mobility in the environment. Copper's movement in soil is determined by a host of physical and chemical interactions with the soil components. In general, copper will adsorb to organic matter, carbonate minerals, clay minerals, or hydrous iron and manganese oxides. Sandy soils with low pHs have the greatest potential for leaching. When the amount of organic matter is low, the mineral content or Fe, Mn, and Al oxides become important in determining the adsorption of copper. Copper binds to soil much more strongly than other divalent cations, and the distribution of copper in the soil solution is less affected by pH than other metals (ATSDR, 2004). The solubility of copper in soil tends to increase as the pH decreases.

Copper binds primarily to organic matter in sediment, unless the sediment is organically poor. It also binds to iron oxides. The solubility of copper in sediments tends to increase as the pH of the sediment decreases.

The BCF of copper in fish obtained in field studies ranges from 10 to 667, indicating a low potential for bioconcentration. The BCF is higher in mollusks, where it may reach 30,000 (Perwak, et al., 1980). This may be due to the fact that many mollusks are filter feeders, and copper concentrations are higher in particulates than in water. There is abundant evidence, however, that there is no biomagnification of copper in the food chain. No evidence of bioaccumulation in herbivorous, omnivorous, and carnivorous mammals was obtained during a study of 10 mammal species in Donana National Park in Spain (Hernandez et al., 1985). A study of metals in cottontail rabbits showed that while the concentration of copper in surface soil was 130 percent higher than in control areas, the concentration of copper in foliar samples was insignificant. No significant increase in copper was observed in rabbit muscle, femur, kidney, or liver, indicating that copper was not bioaccumulating in the food chain. Even at the lowest levels of the food chain, there is little evidence of copper bioaccumulation. In a study of earthworms and soil from 20 different sites, copper concentrations in earthworms poorly correlated with copper in soil (ATSDR, 2004).

At the pH values and carbonate concentrations characteristic of natural waters, most dissolved copper exists as carbonate complexes rather than as free (hydrated) cupric ions. The concentration of dissolved copper depends on factors such as pH, oxidation-reduction potential, and the presence of competing cations (Ca^{2+}, Fe^{2+}, Mg^{2+}, etc.), anions of insoluble cupric salts (OH^-, S^{2-}, PO_4^{3-}, etc.), and organic and inorganic complexing agents. Allard (1995) reported that copper can exist in the form of freely-dissolved divalent copper cation at a pH of less than 6. Complexation of copper with humic acids can increase the mobility of copper in groundwater and/or surface water but will also reduce the bioavailability to biota. The most significant precipitate formed in natural waters is malachite [$Cu_2(OH)_2CO_3$]. As a result of the aforementioned physico-chemical processes, copper in water may be dissolved or associated with colloidal or particulate matter. Copper complexed in colloidal or particulate forms is generally non-mobile. The combined processes of complexation, adsorption, and precipitation control the level of free copper. The chemical conditions in most natural waters are such that, even at relatively large copper concentrations, these processes will reduce the free copper concentration to extremely low values (ATSDR, 2004).

Between pH 5 and 6, adsorption is the principal process for removing copper from water; above pH 6, precipitation becomes more dominant. Copper binding in soil is correlated with pH, cation exchange capacity, organic content of the soil, and presence of iron oxides. Copper may also be incorporated into mineral lattices where it is unlikely to have ecological significance. In soils with high organic carbon content, copper will be tightly bound to organic matter (ATSDR, 2004). The soil/water partition coefficient for copper has been measured to be >64 for mineral soils and >273 for organic soils, indicating a relatively strong affinity for copper to remain adsorbed to soil (ATSDR, 2004). In sediment, copper is generally associated with mineral matter or tightly bound to organic material (Kennish, 1998).

4.4.4 Lead

The chemistry of lead in aqueous solution is highly complex because this element can be found in a multiplicity of forms. The form of lead at any given site is very important since its bioavailability and uptake dynamics are generally dictated by its form. For example, lead fumes, as from a smelter or gasses generated from the discharge of artillery or bullets, are more bioavailable than mining wastes or intact pieces of lead fragments. The difference is therefore not only the size of the particles but its chemical form. It should also be noted that lead in soil can slowly undergo speciation to more insoluble sulfate, sulfide, oxide, and phosphate salts (National Library of Medicine, 1996). Lead has a tendency to form compounds of low solubility with the major anions of natural water. In the natural environment, the divalent form is the stable ionic species of lead. Hydroxide, carbonate, sulfide and sulfate may act as solubility controls in precipitating lead from water. The amount of lead that remains in solution depends upon the pH of the water and the dissolved salt content. Lead is more soluble in softer water and low pH water (ATSDR, 2005a). Complexation of lead with humic acids can increase the mobility of lead in groundwater and/or surface water but will also reduce the bioavailability to biota.

A significant fraction of lead carried by surface water is expected to be in an undissolved form, which can consist of colloidal particles or lead compounds incorporated in other components of surface particulate matter from runoff. Lead may occur as sorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living or nonliving organic matter in water. The ratio of lead in suspended solids to lead in dissolved form ranges from 4:1 to 27:1 (ATSDR, 2005a).

Most lead in soil is retained there and very little is transported into surface water or groundwater (ATSDR, 2005a). The fate of lead in soil is affected by the adsorption at mineral surfaces, the precipitation of sparingly soluble solid forms of the compound, and the formation of relatively stable organic-metal complexes or chelates with soil organic matter. The mobility of lead increases in environments having low pH due to the enhanced solubility of lead under acidic conditions (ATSDR, 2005a). Lead may be immobilized by ion exchange with hydrous oxides or clays or by chelation with humic or fulvic acids in soil (Olson and Skogerboe, 1975). The downward movement of elemental lead and inorganic lead compounds from soil to groundwater by leaching is very slow under most natural conditions except for highly acidic situations (NSF, 1977). The conditions that induce leaching are the presence of lead in soil at concentrations that either approach or exceed the cation exchange capacity of the soil, the presence of materials in soil that are capable of forming soluble chelates with lead, and a decrease in the pH of the leaching solution (e.g., acid rain) (NSF, 1977).

Plants and animals may bioconcentrate lead, but biomagnification is not expected. Although the bioavailability of lead in soil to plants is limited because of the strong adsorption of lead to soil organic matter, the bioavailability increases as the pH and the organic matter content of the soil are reduced. Lead may be taken up in edible plants from the soil via the root system, by direct foliar uptake and translocation within the plant, and by surface deposition of particulate matter. The amount of lead in soil that is bioavailable to most plants depends on factors such as cation exchange capacity, pH, amount of organic matter present, soil moisture content, and type of amendments added to the soil (ATSDR, 2005a). Low alkalinity and low pH conditions in soils can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower trophic level animals, and plants (Jenkins, 1981).

Most lead does not appear to significantly bioaccumulate in most fish. However, bioaccumulation of tetraethyl lead can occur in aquatic organisms (ATSDR, 2005a). Plants commonly take up lead from soil and, therefore, may return it upon decomposition. Because the bioavailability of lead is dependent upon site-specific conditions, the accuracy of the ecological assessment of lead depends heavily on site-specific tests of bioavailability and subsequent toxicity and accumulation.

4.4.5 Zinc

Zinc occurs in the environment mainly in the +2 oxidation state. Sorption is the dominant reaction, resulting in the enrichment of zinc in suspended and bed sediments. Zinc in aerobic waters is partitioned into sediment through sorption onto hydrous iron and manganese oxides, clay minerals, and organic material. The efficiency of these materials in removing zinc from solution varies according to their concentrations, pH, redox potential, nature and concentration of complexing ligands, cation exchange capacity, and the concentration of zinc (ATSDR, 2005b). Similar to copper, zinc is complexed at high pHs and can exist as freely-dissolved divalent cations at lower pHs, thus enhancing its bioavailability. Therefore, as the pH of the water decreases, the concentration of zinc ions in the water phase increases at the same rate as that of the release of zinc from the sediment. In anaerobic environments and in the presence of sulfide ions, precipitation of zinc sulfide limits the mobility of zinc.

In general, zinc sorbs strongly onto soil particles. The mobility of zinc in soil depends on the solubility of the speciated forms of the element and on soil properties such as cation exchange capacity, pH, redox potential, and chemical species present in the soil; under anaerobic conditions, zinc sulfide is the controlling species (Kalbasi, et al., 1978). Since zinc sulfide is insoluble, the mobility of zinc in anaerobic soil is low. The mobility of zinc in soil increases at lower soil pH under oxidizing conditions and at lower cation exchange capacity of soil (Tyler and McBride, 1982). Distribution constants for zinc in soil range widely from 0.1 to 8,000 L/kg (Baes and Sharp, 1983). Zinc in soluble form (e.g. zinc sulfate) is moderately mobile in most soils; however, the mobility is limited by a slow rate of dissolution. Consequently, movement towards groundwater is expected to be slow unless the zinc in the soil is in the soluble form or is accompanied by corrosive substances (e.g. mine tailings).

Zinc is an essential nutrient that is present in all organisms. Although biota appears to be a minor reservoir of zinc relative to soils and sediments, microbial decomposition of biota in water can produce ligands, such as humic acids, that can affect the mobility of zinc in the aquatic environment through zinc precipitation and adsorption (ATSDR, 2005b). Zinc can accumulate in freshwater animals at 51 to 1,130 times the concentration present in water (USEPA, 1987). In general, zinc does not biomagnify through food chains. Furthermore, although zinc bioaccumulates to some degree in aquatic systems, biota appears to represent a relatively minor sink compared to sediments. Steady-state zinc BCF for 12 aquatic species range from 4 to 24,000, with most being less than 100 (USEPA, 1987). With respect to bioconcentration from soil by terrestrial plants, invertebrates, and mammals, BCFs of 0.4, 8, and 0.6, have been reported, respectively. In general, plants do not concentrate zinc above levels present in the soil (ATSDR, 2005b).

4.4.6 Gamma-Chlordane

Technical chlordane is a mixture of some 50 different compounds; the major constituents being alpha-, beta, and gamma-chlordane (Howard, 1991). Chlordane is generally immobile in soil; however, movement to groundwater can occur (Howard, 1991). Chlordane is extremely persistent in the environment and may persist in some soils for over 20 years. Volatilization appears to be the only major removal mechanism from soil; however, leaching to groundwater may occur. Adsorption to sediments and volatilization are important removal mechanisms in water. In air, chlordane exists predominantly in the vapor phase. Vapor-phase chlordane degrades by photolysis and hydroxyl radical reaction (ATSDR, 1994b).

Chlordane in water will adsorb to bed and suspended sediments and volatilize. The partitioning of chlordane to sediment correlates with the organic carbon content of the sediment. The rate of volatilization of chlordane from water depends to a large extent on the amount, size, and composition (i.e., percent organic matter) of the suspended material in the water body since adsorption to suspended solids and sediments attenuates the rate of volatilization (Oloffs et al., 1973). Chlordane will bind tenaciously to dissolved organic carbon in water which will result in increased apparent solubility and mobility in the presence of dissolved organic carbon. Chlordane volatilizes rapidly from water, and it appears that volatilization kinetics may be faster than adsorption kinetics. The majority of chlordane, however, probably enters water as runoff from urban and agricultural soils and is adsorbed to particulates before entering a water body. The chlordane repartitions in the water and volatilizes rapidly near the water surface (ATSDR, 1994b).

Chlordane will bioconcentrate in both marine and freshwater species. Marine bioconcentration factors have been measured as high as 3,000 to 12,000 (Zaroogian et al., 1985) and freshwater bioconcentration factors have been measured in rainbow trout as high as 18,500 (Oliver and Niimi, 1985). Chlordane may also biomagnify through the food chain. Chlordane is also taken up by rooted aquatic vascular plants from both water and sediment.

In soil, chlordane adsorbs to the organic matter and volatilizes slowly over time. Chlordane does not leach significantly. In general, chlordane remains in the top 20 centimeters of most soils and for some soils it stays at this level for over 20 years (Beeman and Matsumura, 1981). Volatilization from soil is a major loss mechanism for chlordane. The rate of volatilization depends on such parameters as the soil organic content, water content, temperature, and relative humidity as well as its vapor pressure and adsorption to soil. In general, sandy soils, and soils with small amounts of organic matter retain chlordane less than soils with high clay and/or organic content (Wiese and Basson, 1966). Soil moisture, however, is the most important factor governing volatilization rates from soil.

5.0 Ecosystems Potentially at Risk

In general, the ecological habitats present at the BBGR Ranges can be categorized into upland and aquatic habitat types. The upland habitats can be described as either "cleared" areas or forested areas. The cleared areas, or old field habitat, are those areas that were formerly maintained as lawns or mowed fields. These areas represent the locations where range activities were most prevalent. Since maintenance activities have ceased in these areas, pioneer species are colonizing these areas. Typically, the species most likely to colonize these areas are the "weed" species that tend to be vigorous pioneer plants that grow and spread rapidly. The first of the pioneer species to invade these abandoned areas are the grasses and herbaceous species. These formerly maintained grassy areas are classified as being in an early old field successional state. Over time, these grass and herbaceous species will be followed by shrubs and small trees. The early old field successional areas at the BBGR Ranges are dominated by various grasses and herbs including dock (*Rumex spp.*), clover (*Trifolium spp.*), vetch (*Astragalus spp.*), milkweed (*Ascelepias spp.*), bed straw (*Galium spp.*), ox-eye daisy (*Chrysanthemum leucanthemum*), and Johnson grass (*Sorghum halepense*). Other old field herbaceous species occurring at the BBGR Ranges are black raspberry (*Rubus occidentalis*), poison ivy (*Toxicodendron radicans*), smooth sumac (*Rubus glabra*), green brier (*Smilax rotundiflora*), Japanese honeysuckle (*Lonicera japonica*), fox grape (*Vitus labrusca*), and multiflora rose (*Rosa multiflora*). Loblolly pine (*Pinus taeda),* longleaf pine *(Pinus palustris),* and shortleaf pine (*Pinus echinata*) saplings have also begun to encroach on the formerly maintained grassy areas of the BBGR Ranges.

The forested areas outside of the cleared areas are best characterized as mixed deciduous/coniferous forest. The canopy species typically found in the forested areas surrounding the BBGR Ranges include yellow poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), white oak (*Quercus alba*), and northern red oak *(Quercus rubra*). The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry *(Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this habitat type.

The aquatic habitats in the vicinity of the BBGR Ranges can be divided into two separate drainages: Ingram Creek tributaries and South Branch of Cane Creek tributaries. The Ingram Creek tributaries drain Ranges 20, 23, 25, 25-East, 26, and Ranges South of Range 25. The South Branch of Cane Creek tributaries drain Range 18. All of these tributaries are small ephemeral streams that normally only exhibit flowing water during periods of heavy precipitation, with the exception of the small tributary that flows from east-to-west across Range 23. This small tributary can be considered perennial as it contains water throughout the majority of the year under normal precipitation conditions. The tributaries in both of these drainages are generally narrow (less than 3 feet wide) and shallow when water is present (less than 6 inches deep). The substrate is mostly sand and gravel with significant amounts of leaf litter within the forested areas.

Terrestrial species that may inhabit the area of the BBGR Ranges include opossum, short-tailed shrew, raccoon, white-tail deer, red fox, coyote, gray squirrel, striped skunk, a number of species of mice and rats (e.g., white-footed mouse, eastern harvest mouse, cotton mouse, eastern woodrat, and hispid cotton rat), and eastern cottontail. Approximately 200 avian species reside at FTMC at least part of the year (USACE, 1998). Common species expected to occur in the vicinity of the BBGR Ranges include northern cardinal (*Cardinalis cardinalis*), northern mockingbird (*Mimus polyglottus*), warblers *(Dendroica spp.*), indigo bunting (*Passerina cyanea*), red-eyed vireo (*Vireo olivaceus*), American crow (*Corvus brachyrhynchos*), bluejay (*Cyanocitta cristata*), several species of woodpeckers (*Melanerpes spp., Picoices spp.*), and Carolina chickadee (*Parus carolinensis*). Game birds present in the vicinity of the BBGR Ranges may include northern bobwhite (*Colinus virginianus*), mourning dove (*Zenaida macroura*), and eastern wild turkey (*Meleagris gallopavo*). Woodland hawks (e.g., sharp-shinned hawk) were observed in this area during the ecological investigation (December 2002) and are expected to use this area for a hunting ground. A variety of other raptors (e.g., red-tailed hawk, barred owl, and great horned owl) could also use portions of this area for a hunting ground, particularly the fringe areas where the forested areas abut roads and cleared areas. Although the small size and ephemeral nature of the tributaries at the BBGR Ranges limits the potential for the occurrence of piscivorous birds, they may be present during limited periods of the year when precipitation is heavy and the tributaries exhibit flowing water. The piscivorous bird species that may be present in the vicinity of the BBGR Ranges include great blue heron (*Ardea herodias*), green-backed heron (*Butorides striatus*), and belted kingfisher (*Ceryle alcyon*).

In general, the terrain at FTMC supports large numbers of amphibians and reptiles. Jacksonville State University has prepared a report titled *Amphibians and Reptiles of Fort McClellan, Calhoun County, Alabama* (Cline and Adams, 1997). The report indicated that surveys in 1997 found 16 species of toads and frogs, 12 species of salamanders, 5 species of lizards, 7 species of turtles, and 17 species of snakes. Typical inhabitants of the area surrounding the BBGR Ranges are copperhead (*Agkistrodon contortix*), king snake (*Lampropeltis getulus*), black racer (*Coluber constrictor*), fence lizard (*Sceloporour undulatus*), and six-lined racerunner (*Cnemidophorous sexlineatus*).

Descriptions of the habitats at each of the BBGR Ranges are presented in the following sections.

5.1 Range 18 Habitat

Range 18 is comprised of two main habitat types: cleared and forested areas. The cleared area comprises the vast majority of Range 18. The entire area of Range 18, including the extensive safety fan is approximately 3,304 acres. The study area of Range 18 is approximately 70 acres in size. The overall elevation of Range 18 ranges from approximately 800 to 875 feet above mean sea level (msl), with the highest elevation at the top of the hill which forms the impact zone in the southern portion of the study area. Ground surface is flat in the firing line area with a slight downward slope to the northeast.

The study area of Range 18 is almost entirely comprised of formerly maintained lawn, mowed fields, and unvegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine (*Pinus taeda*) saplings and shortleaf pine (*Pinus echinata*), have also begun to encroach into these previously maintained areas. Significant portions of Range 18 remain unvegetated, with large areas of bare soil.

The forested areas surrounding Range 18 are best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally to the north and east from the impact zone area via several ephemeral creeks and ditches and transects the range immediately behind the 75-meter target line. This water eventually connects to South Branch of Cane Creek northeast of Range 18, flowing further north towards Cane Creek. These ephemeral ditches are generally narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these ditches is mostly sand and gravel with a significant amount of leaf litter present where the ditches run through forested areas. The vegetation within these ditches (when present) is characteristic of the upland habitats surrounding the ditches and is not characteristic of wetland or aquatic habitats, indicating that these ditches only carry water during significant rainfall events. During the majority of the year, these ditches are characteristic of the upland habitats present at the site.

The ephemeral drainage features that flow through Range 18 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). These drainage features have been identified as potential gray bat foraging habitat because they provide habitat for aquatic insects, which are fed upon by the gray bat. However, these drainage features contain water only during periods of significant precipitation and are dry during the majority of the year. Thus, aquatic insects would not be expected to occur in these ditches for the majority of the year. Additionally, gray bats prefer continuous cover while traveling to and from their foraging habitats and while foraging. Due to historical maintenance activities at Range 18, the forest canopy has been eliminated and only grasses and weeds remain over the majority of Range 18. Thus, the currently existing vegetation at Range 18 does not provide the cover favored by gray bats. Studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3D/International [3DI], 1997).

5.2 Range 20 Habitat

Range 20 is comprised of two main habitat types: cleared and forested areas. The study area (33.4 acres) extends over the entire range including the M-60 impact zone area (1.7 acres). The range safety fan extends to the southeast, covering an area of approximately 1,505 acres.

A small, ephemeral tributary of Ingram Creek cuts across Range 20 draining surface water runoff west towards Ingram Creek. The overall elevation of Range 20 ranges from approximately 925 to 1,200 feet above msl. The lowest elevation is found near the Ingram Creek tributary and the highest elevation is at the top of the hill in the impact zone. Ground surface is relatively flat in the M-60 firing point and explosive pit area with a 75 foot decent to the tributary in the infiltration training area and a steep 1,025 foot rise towards the impact zone hillside southeast of the M-60 firing points.

The study area of Range 20 is almost entirely forested, with the exception of the firing point and the impact zone. The former firing point area has been re-graded and is entirely enclosed with chain-link fence, and is currently used by the Corps of Engineers for storage. The impact zone exhibits several areas of bare, unvegetated ground and the remaining portion is forested with immature (less than 5 years old) mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this area as signs of fire indicate that these vegetative layers have been burned in recent years. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally to the west-southwest from the firing point and impact zone areas via a small, ephemeral stream that bisects the firing point and impact areas. This tributary eventually connects to Ingram Creek west of Range 20. This ephemeral ditch is generally narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in this ditch is mostly sand and gravel with a significant amount of leaf litter.

Ingram Creek and its tributaries in the vicinity of Range 20 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the ephemeral nature of the small tributary at Range 20, aquatic insects would not be expected to occur in this area for significant periods during any given year. Additionally, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.3 Range 23 Habitat

The study area where firing points, lanes, and target mounds are located is defined as approximately 58.6 acres. The range safety fan extends to the southeast and covers an area of approximately 4,566 acres.

Snap Lane forms the western boundary of the Range 23 study area. Three tributaries of Ingram Creek are present at the range, carrying runoff water from the hillside east of the range, through the firing lane area, under Snap Lane, and northwards towards Ingram Creek. Two of these tributaries are ephemeral, while the tributary that flows from the central portion of Range 23 generally contains water throughout the year. The elevation of Range 23 increases from approximately 850 to 900 feet above msl at the firing lane area in the western portion of the range, steeply rising beyond the 300-meter target mounds to an elevation of 1,050 feet at the top of the unnamed hill east of the main study area.

Range 23 exhibits both cleared and forested areas. The 16 firing lanes are characteristic of cleared areas. The firing lanes are separated from each other by narrow strips of forest. The formerly cleared firing lanes are characteristic of old field, early successional habitat. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these areas. There are significant areas within the firing lanes that are unvegetated, with large areas of bare soil. The narrow strips of forest between the firing lanes can be characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. These forested strips also exhibit dense understory and herbaceous layers. The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Surface water drainage is generally from east-to-west across Range 23 via three small creeks/ditches. Two of these creeks (the northern-most and southern-most) are ephemeral in nature, while the third creek contains water throughout the majority of the year. All three of these creeks eventually connect to Ingram Creek west of Range 23. All three of these creeks are narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these creeks is mostly sand and gravel. Vegetation in the two ephemeral tributaries is characteristic of the upland vegetation that occurs in the surrounding area, indicating that these tributaries do not contain water for significant portions of the year.

Ingram Creek and its tributaries in the vicinity of Range 23 have been identified as low-quality foraging habitat for the Federally-listed endangered gray bat (*Myotis grisescens*) (Garland, 1996). Ingram Creek and its tributaries have been identified as a potential gray bat foraging area because they provide habitat for aquatic insects, which are fed upon by the gray bat. Due to the ephemeral nature of two of the tributaries at Range 23, aquatic insects would not be expected to occur in these tributaries for significant periods during any given year. Aquatic insects may be present in the central tributary since it is more perennial in nature. However, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.4 Range 25 Habitat

Range 25 is comprised of two main habitat types: cleared and forested areas. The cleared area comprises the vast majority of Range 25. The total area of Range 25, including the extensive safety fan, comprises 1,713 acres. The main study area of Range 25 is approximately 103 acres and is topographically relatively flat. A hillside in the northern portion of the study area forms one of the major impact zones for this range. The study area of Range 25 is comprised almost entirely of formerly maintained lawns, mowed fields, and unvegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. Significant portions of Range 25 remain unvegetated, with large areas of bare soil.

The forested area in the northern portion of Range 25 is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Three small, ephemeral tributaries to Ingram Creek and Cane Creek flow from east-to-west across the study area of Range 25. The southern-most tributary and the tributary that crosses through the center of the study area drain to Ingram Creek. The northern-most tributary crosses the study area in the northwest corner of the study area and leads to Cane Creek to the northwest. All three of these tributaries are narrow (< 3 feet wide) and shallow (< 6 inches deep) when water is present. The substrate in these creeks is mostly sand and gravel except where they traverse wooded areas, in which case they have significant leaf litter. Vegetation in these ephemeral tributaries is characteristic of the upland vegetation that occurs in the surrounding area, indicating that these tributaries do not contain water for significant portions of the year.

The small ephemeral tributaries that cross Range 25 are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.5 Range 25-East Habitat

It is not certain whether Range 25 East was ever constructed. The study area for Range 25 East encompasses the area where possible firing lines and impact zones are located and is approximately 44 acres. There is no safety fan associated with Range 25 East. The other ranges which overlap this study area include Range 20 and Range 26. If it was constructed in the same layout and orientation as Range 25, Range 25 East would have featured firing lines to the south (in the vicinity of the Range 26 firing line) and a target line berm to the northeast. The hillside that would act as a possible impact zone for this range rises approximately 150 feet in elevation in the northern portion of the study area downrange of the original first firing line. The elevation of Range 25 East ranges from approximately 825 feet (in the southern portion of the study area) to 1,025 feet above msl at the top of the hillside in the possible impact zone. Ground surface rises gradually in the firing line area with a steep rise in the possible impact zone.

The majority of Range 25 East is forested with mixed deciduous/coniferous forest. The southeastern corner of the study area (which overlaps the firing line area of Range 26) is characteristic of old field, early successional habitat. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate the forest habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. The old field, early successional portion of the study area is dominated by various grasses and herbaceous species that have grown uncontrolled since maintenance activities have ceased. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Range 25 East is bisected by Bains Gap Road, which runs through the impact zone area in the northern portion of the range. A small ephemeral tributary of Ingram Creek drains runoff water from the hillside in the northern portion of the study area (where the possible impact zone would be). Two other ephemeral tributaries (discussed in Range 20 and Range 26 sections) drain runoff from the east to the west across the study area of Range 25 East. These tributaries meet near the parking area at Range 26, cross under Bains Gap Road, and continue flowing west towards Ingram Creek. The small ephemeral tributaries that cross Range 25 East are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.6 Range 26 Habitat

The study area for Range 26 (where firing lanes and impact zones are located) is approximately 34.2 acres. The range safety fan extends to the southeast covering an area of approximately 1,267 acres. The overall elevation of the Range 26 study area gradually increases from approximately 875 to 900 feet above msl in the western portion of the study area to approximately 900 to 985 feet above msl at the hillside impact zone in the eastern portion of the study area.

The study area of Range 26 is comprised almost entirely of formerly maintained lawns, mowed fields, and unvegetated soil. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Loblolly pine saplings (*Pinus taeda*) have also begun to encroach into these previously maintained areas. Significant portions of Range 26 remain unvegetated, with large areas of bare soil.

On the periphery of the study area the habitat is best characterized as mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

Two ephemeral tributaries of Ingram Creek collect runoff from the hillsides to the south and northeast of the study area and traverse the study area from east-to-west. Both of these drainage features exhibit vegetation that is characteristic of the surrounding upland vegetation, indicating that these ditches only transmit water during periods of significant precipitation. Additionally, these small ephemeral tributaries that cross Range 26 are not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.7 Ranges South of Range 25 Habitat

The Ranges South of Range 25 encompass an area of approximately 6.1 acres and includes portions of several historical ranges, including Parcels 224Q, 226Q, and 227Q. The study area is bounded on the east by Snap Lane and on the north by Bains Gap Road. There are no safety fans associated with the Ranges South of Range 25. The study area is relatively flat with a total elevation change of approximately 25 feet, sloping from east-to-west.

The entire study area is forested with a somewhat immature mixed deciduous/coniferous forest. Shortleaf pine, loblolly pine, white oak, and southern red oak dominate the forest habitat. The study area exhibits dense understory and herbaceous layers. The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by mountain laurel (*Kalmia latifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), Christmas fern (*Lystrichum acrotichoides*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this area. White-tailed deer, wild turkey, gray squirrel, and various song birds have been observed on-site.

A small ephemeral tributary of Ingram Creek runs through the center of the study area from east-to-west. This drainage feature exhibits vegetation that is characteristic of the surrounding upland vegetation, indicating that this ditch only transmits water during periods of significant precipitation. The small ephemeral tributary that crosses the study area is not classified as potential gray bat foraging habitat. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.8 Ingram Creek Tributaries Habitat

The tributaries to Ingram Creek that are present at the BBGR Ranges drain Ranges 20, 23, 25, 25-East, 26, and Ranges South of Range 25. All of these tributaries are small ephemeral streams with the exception of one of the tributaries that flows across Range 23, which is perennial. Surface water runoff from the hills located east and north of the BBGR Ranges is the major source of water for these tributaries. As such, they only convey water during periods of significant precipitation. Flow in these small ephemeral tributaries is highly variable, depending on precipitation in the surrounding watershed. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution from groundwater varies according to the amount of precipitation, with an increase when precipitation raises the potentiometric surface. Downstream (west) of the BBGR Ranges, these small ephemeral tributaries continue to flow in a westerly direction towards Ingram Creek. Ingram Creek (located west of the BBGR Ranges) flows in a northwesterly direction until its confluence with Cane Creek, northwest of the BBGR Ranges.

The headwater areas for these tributaries are relatively undeveloped portions of the Main Post and are almost entirely mixed deciduous/coniferous forest. The physical characteristics of these tributaries at the BBGR Ranges are relatively consistent. All of these tributaries are narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in these tributaries where they pass through forested areas. The vegetation in these tributaries is generally characteristic of the surrounding upland vegetation, indicating that these tributaries only transmit water during periods of significant precipitation.

Although the Ingram Creek tributaries at the BBGR Ranges are ephemeral in nature, they have the potential to support a variety of amphibious species and some small, drought-tolerant fish species. Bullfrog *(Rana catesbeiana)* and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in these small ephemeral tributaries in the vicinity of the BBGR Ranges. Fish species that may be found in the Ingram Creek tributaries in the vicinity of the BBGR Ranges include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*), and various darters (*Etheostoma spp.*). The shallow nature of these ephemeral tributaries limits their ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Larger fish species are not expected to inhabit the Ingram Creek tributaries in the vicinity of the BBGR Ranges because they only hold water during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

The small ephemeral tributaries to Ingram Creek in the vicinity of the BBGR Ranges are either not classified with regard to potential gray bat foraging habitat or are classified as low quality gray bat foraging habitat. The tributaries that are classified as low quality gray bat foraging habitat are classified as such because they provide habitat for aquatic insects, which may be fed upon by the gray bat. However, these drainage features contain water only during periods of significant precipitation and are dry during the majority of the year. Thus, aquatic insects would not be expected to occur in these ditches for the majority of the year. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).

5.9 South Branch of Cane Creek Tributaries Habitat

A single tributary to South Branch of Cane Creek occurs within the BBGR Ranges and only at Range 18. This small, ephemeral tributary runs along the western boundary of Range 18 and then transects the range from west-to-east in the northern portion of the range. This small tributary then turns north along the eastern boundary of Range 18 and joins South Branch of Cane Creek immediately north of the Range 18 study area. Surface water runoff from the hills west and south of Range 18 is the major source of water for this tributary. As such, it only conveys water during periods of significant precipitation. There also appears to be localized contribution to tributary flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution from groundwater varies according to the amount of precipitation, with an increase when precipitation raises the potentiometric surface.

The headwater areas for this tributary (immediately south and west of Range 18) are relatively undeveloped portions of the Main Post and are almost entirely mixed deciduous/coniferous forest. The physical characteristics of this tributary are consistent with other small ephemeral drainage features across FTMC. This tributary is narrow (< 3 feet wide) and shallow (< 6 inches deep), when water is present, and exhibit a substrate of mostly sand and gravel. There is significant leaf litter present in this tributary where it passes through forested areas. The vegetation within this tributary is generally characteristic of the surrounding upland vegetation, indicating that this tributary only transmits water during periods of significant precipitation.

Although the South Branch of Cane Creek tributary at the BBGR Ranges is ephemeral in nature, it has the potential to support a variety of amphibious species and some small, drought-tolerant fish species. Bullfrog *(Rana catesbeiana)* and leopard frog (*Rana sphenocephala*) are examples of amphibians that may be found in this small ephemeral tributary. Fish species that may be found in the South Branch tributary at Range 18 include blacknose dace (*Rhinichthys atratulus*), creek chub (*Semotilus atromaculatus*), stoneroller (*Campostoma anomalum*), striped shiner (*Luxilus chrysocephalus*), and various darters (*Etheostoma spp.*). The shallow nature of this ephemeral tributary limits its ability to support many aquatic organisms (e.g. large fish) and other organisms that rely on aquatic species for food (e.g. piscivores). Larger fish species are not expected to inhabit the South Branch tributary in the vicinity of Range 18 because it only has water present during periods of significant precipitation, and when water is present, it is too shallow to support larger fish.

The South Branch of Cane Creek tributary within the Range 18 study area is classified as low quality gray bat foraging habitat. South Branch of Cane Creek, located just east and north of the Range 18 study area, is classified as moderate quality gray bat foraging habitat. The tributary is classified as low quality gray bat foraging habitat because it has the potential to provide habitat for aquatic insects, which may be fed upon by the gray bat. However, this drainage feature contains water only during periods of significant precipitation and is dry during the majority of the year. Thus, aquatic insects would not be expected to occur in this ditch for the majority of the year. Additionally, gray bats prefer continuous cover while traveling to and from their foraging habitats and while foraging. Due to historical maintenance activities at Range 18, the forest canopy has been eliminated and only grasses and weeds remain over the majority of Range 18. Thus, the currently existing vegetation at Range 18 does not provide the cover favored by gray bats. Furthermore, studies conducted to assess the presence of gray bats at FTMC and their home ranges have indicated that gray bats do not use this area as foraging habitat (3DI, 1997).